Economic insights in ecological compensations

Market analysis with an empirical application to the Finnish economy

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Abstract

Biodiversity degrades at an alarming rate, both globally and in Finland. Habitat loss is the most significant threat for biodiversity. Biodiversity offsets (also called ecological compensation) are becoming a common market-based policy instrument, aimed at balancing economic development and conservation of ecosystems and species. Offsets are designed to compensate for the residual environmental impacts of development projects, after avoiding and minimizing impacts on site. The idea is that costs of conservation are allocated to the party responsible for habitat degradation, thus a polluter pays principle is implemented. Offsets complement the pre-existing conservation instruments. Ecological risks as well as the theoretical and practical challenges of offsetting are widely discussed in literature but economic analysis on biodiversity offsetting schemes is limited to few.

The aim of this thesis is to increase the understanding of the economic basis of biodiversity offset markets and in particular, the influence of trading ratios and intermediaries. I developed an equilibrium model, and applied it to Finnish data and three selected habitat types: abundant mires, scarce herb-rich forests, and laborious and valuable rural biotopes. The supply of offsets comes from habitat restoration and nature management. Data on the areas suitable for habitat restoration, restoration measures and associated costs were obtained from several documented sources. I utilized the results of the working group on improving the status of habitats in Finland (ELITE, Kotiaho et al. 2015), and supplemented it with an expert survey that I designed to estimate the changes in the selected habitat types after restoration and management under uncertainties. I used Monte Carlo simulation to examine the impacts and risks of uncertainties. Further, I estimated demand based on a report by Tiitu et al. (2015) where they predict the increase of built-up areas and infrastructure in Finland for a time period of 2013-2040.

I examined how the market equilibrium, prices, and quantities traded depended on trading ratios. Trading ratios differ depending on whether biodiversity losses from development are ecologically equivalent to gains from compensation or not. I also examined the role of an intermediary, a broker firm. The intermediary helps demanders and suppliers meet each other with minimal transaction costs, safeguards against risks and guarantees maturity and quality of offsets. The analysis showed that the presence of the intermediary affects the trading ratios as there is a time delay between losses and gains which must be discounted to present time if the intermediary is not in the market guaranteeing mature offsets. Time discounting further increases trading ratios.

The results show that the market size could be considerable and providing offsets could be a profitable business for landowners. There is enough land for compensations in Finland, even when trading ratios are relatively high. The presence of the intermediary in the market decreases both the trading ratios and credit prices, which lowers the costs of compensation for developers. Both ecological and economic risks may decrease as the intermediary safeguards against failures in restoration by guaranteeing that all offsets provide good quality. Pricing these services in the market does not excessively increase offset prices and shrink the market size.

Keywords

Biodiversity, biodiversity offsets, ecological compensation, offset markets, environmental economics

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Tiivistelmä

Luonnon monimuotoisuus eli biodiversiteetti vähenee huolestuttavaa vauhtia Suomessa ja maailmalla. Elinympäristöjen heikentyminen on suurin uhka biodiversiteetille. Biodiversiteettikompensaatiot ovat yleistymässä maailmalla uutena, markkinaperusteisena biodiversiteetin suojelun ohjauskeinona. Pilaaja maksaa -periaate toteutuu, kun elinympäristöjen heikentäjä korvaa syntyvän haitan, jota ei voida välttää tai minimoida, hankkimalla vastaavan määrän biodiversiteettikompensaatioita. Kompensaatiot eivät vaikuta olemassa oleviin toimiin ja ohjauskeinoihin, kuten suojelualueiden määrään tai perustamiseen.

Tämän tutkielman tavoitteena on lisätä ymmärrystä kompensaatiomarkkinoiden toiminnasta ja tutkia erityisesti vaihtosuhteen ja välittäjäorganisaatioiden vaikutusta markkinatasapainoon. Rakensin tasapainomallin, jota sovellan suomalaiseen dataan ja kolmeen esimerkkiluontotyyppiin: tavanomainen suoluonto, uhatut lehdot sekä jatkuvaa hoitoa vaativat arvokkaat perinnebiotoopit. Kompensaatioiden tarjonta syntyy elinympäristöjen ennallistamisesta ja luonnonhoidosta sekä suojelusta. Yhdistin aineistoa ennallistamisen ekologisista vaikutuksista ja kustannuksista sekä soveltuvien kohteiden pinta-aloista useista eri julkaisuista. Hyödynsin erityisesti Elinympäristöjen tilan edistäminen Suomessa (ELITE) -raportin tietoja ja täydensin niitä suunnittelemalla ja toteuttamalla asiantuntijoille kyselyn, jolla kartoitin luontotyyppien ekologisen tilan kehityssuuntia ja epävarmuutta. Monte Carlo -simuloinnin avulla selvitin ennallistamisen epävarmuuden vaikutusta ja riskejä. Kompensaatioiden kysyntäpaineen estimoin rakennetun alueen muutosten ennusteiden eli yhdyskuntarakenteen ja infrastruktuurin kasvuennusteiden avulla. Otin huomioon myös turvetuotantoalan kasvun ja biodiversiteettiä epäsuorasti heikentävien yritysten kysynnän.

Mallin avulla tutkin, kuinka markkinatasapaino eli kompensaatioiden hinnat ja vaihdetut määrät riippuvat vaihtosuhteesta. Vaihtosuhde määrittää, kuinka paljon kompensaatioita on ostettava suhteessa aiheutettuun haittaan. Vaihtosuhde vaihtelee riippuen siitä, ovatko yritysten aiheuttamat biodiversiteettihaitat ja kompensaatioiden avulla tuotettavat biodiversiteettiparannukset yhtä suuria. Lisäksi analysoin välittäjäorganisaation roolia markkinoilla. Välittäjä sujuvoittaa vaihtoa kysyjien ja tarjoajien välillä ja ylläpitää portfoliota valmiista kompensaatioista. Taloudelliset ja ekologiset riskit pienenevät, kun välittäjä takaa valmiit kompensaatiot ja kantaa riskin ennallistamisen epäonnistumisesta. Välittäjän puuttuessa haitan ja parannuksen välillä on aikaviive, joka diskontataan nykyaikaan, mikä nostaa vaihtosuhdetta.

Tulokset osoittavat, että kompensaatiomarkkinoiden potentiaalinen koko on kohtuullisen suuri. Kompensaatioiden tuottaminen voi tulosten perusteella olla kannattavaa liiketoimintaa maanomistajille. Suomessa riittää maa-alaa kompensaatioihin, vaikka vaihtosuhde nousisi moninkertaiseksi suhteessa haittaan. Kun markkinoilla on mukana välittäjä, kompensaatioiden vaihtosuhde ja hinta laskevat, jolloin kompensaatiotoiminta on yrityksille edullisempaa.

Yhteenvetona voidaan todeta, että Suomessa on potentiaalia ekologisten kompensaatioiden tuottamiseen, koska luontotyyppien ennallistamisesta on runsaasti kokemusta ja ennallistettavaksi sopivia kohteita on suhteellisen paljon. Suomessa on myös kysyntäpotentiaalia, joka syntyy maankäytön muutoksista sekä yritysten halukkuudesta hankkia kompensaatioita myös ilman suoria vaikutuksia biodiversiteettiin.

Avainsanat

Biodiversiteetti, ekologiset kompensaatiot, kompensaatiomarkkinat, ympäristöekonomia

Säilytyspaikka

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Muita tietoja

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1. Introduction

Biodiversity is decreasing at an alarming rate. Increasing human population, land-use and consumption cause ecosystem degradation and are among the greatest threats for biodiversity and ecosystem services, as well as for the future of humanity (MEA 2005, 67). Everyone depends on Earth's ecosystems and the services they provide. Biodiversity is an essential underlying feature of well-functioning ecosystems that provide numerous ecosystem services: clean water, food, raw materials, nutrient recycling, pollination, climate regulation and recreation, to mention a few. Human use of ecosystem services is growing rapidly: approximately 60 % of the ecosystem services evaluated in the Millennium Ecosystem Assessment are being degraded or used unsustainably. For instance, half of provisioning services and 70 % of regulating and cultural services are being degraded. (MEA 2005, 39-47.)

Globally, at least 10-30 % of mammal, bird, and amphibian species are currently threatened with extinction with medium to high certainty (65-98% probability). The average rate of extinction found for marine and mammal fossil species is approximately 0.1-1 extinctions per million species per year. Approximately 100 species of birds, mammal, and amphibians have become extinct over the past 100 years, which is 50-500 times the background rates. When possibly extinct species are included, humans have increased the species extinction rate by as much as 1000 times the background rates. (MEA 2005, 35-36.)

Also in Finland, the state of biodiversity continues to worsen (Rassi et al. 2010, 12-13, Raunio et al. 2008a). In the red list assessment of Finnish species in 2000 and 2010, classification of 66 % of the assessed species has changed towards more threatened red-list classes (Rassi et al. 2010, 125-134). Forests are the most important habitat for red-listed species: approximately 30 % of red-listed species suffer primarily from changes in the forest environment (Rassi et al. 2010, 49-51). Decreasing amount of decaying wood, forest management activities, changes in the tree species composition of forests, the reduction of old-growth forests, and decreasing number of large trees is a significant threat to forest species. Also, reduction of burnt forest areas and other young stages of natural succession cause threat (Rassi et al. 2010, 61-66). In addition, the overgrowing of meadows and other open habitats, as well as construction of buildings, infrastructure and waterways, mining, peatland drainage for forestry and peat harvesting threaten biodiversity in Finland. (Rassi et al. 2010, 49-51.)

There is political will to cooperate internationally to stop biodiversity loss. Several global and European Union (EU) strategies and agreements have been set to tackle biodiversity loss. The Convention on Biological Diversity (CBD) steers the conservation and sustainable use of biodiversity. The main objective is to halt the degradation of biodiversity by 2020 at global, regional and national scales (CBD 2010). Strategic Plan for Biodiversity 2011–2020 is an international framework implemented by the parties of the Convention. The plan comprises five strategic goals and 20 so-called Aichi targets. Strategic goals are to address the underlying causes of biodiversity loss, to reduce the direct pressures on biodiversity, to improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity, to enhance the benefits to all from biodiversity and ecosystem services and to enhance implementation through participatory planning, knowledge management and capacity building.

The EU biodiversity strategy has a headline target of "Halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss" (European Commission 2011). In Finland, the national strategy and action plan for the conservation and sustainable use of biodiversity steers nationwide species protection and implements the decisions of both the CBD Strategic Plan and the objectives of the EU Biodiversity Strategy. (Ministry of the Environment 2012.)

The Ministry of the Environment of Finland provides funding for the protection and management of threatened species but the needs for conservation activities consistently exceed the available budgets (Ministry of the Environment 2012; Rassi et al. 2010, 144). Despite the implemented agreements and current efforts, biodiversity continues to degrade in Finland – albeit at a slower rate than before (Rassi et al. 2010, 125–134; Raunio et al. 2008a; Raunio et al. 2013). As current actions are not sufficient, there is a need for new policy tools to halt the ongoing loss of biodiversity.

The international strategies and agreements mentioned above consider biodiversity offsets as a tool for reaching their objectives. Offsets are becoming a common market-based policy instrument, aimed at balancing economic development and the conservation of ecosystems and species (OECD 2016). In accordance with a so-called mitigation hierarchy, offsets seek to compensate only for the unavoidable, residual environmental impacts of project development, after avoiding and minimizing impacts on site (McKenney & Kiesecker 2010). They complement the existing conservation measures, allocating the costs of conservation to the party responsible for habitat degradation, in accordance with a polluter pays principle.

From an economic angle, biodiversity is a public good. This entails that no-one can be excluded from using the good and one's consumption does not reduce the availability of the good for others (Hanley et al. 2007, 39). Public goods are typically undersupplied, as market prices of land and raw materials do not signal biodiversity provision or changes in it. Therefore, there are no incentives for developers to consider biodiversity in their decision making. This leads to economic activity causing a negative externality to the environment: production and consumption cause damage which is not compensated in the price. Typically, this market-failure has been corrected with government intervention and biodiversity conservation has been steered with regulation and protected areas. Biodiversity offsets are designed to engage those actors that cause degradation and to internalize the external costs of development projects.

The aim of this thesis is to increase the understanding of the economic basis of biodiversity offset markets and in particular, the influence of trading ratios and intermediaries. I develop an equilibrium model to study a specific mechanism to implement offsetting, habitat banking. In general, the term habitat bank refers to a restored land area where credits are sold. Thus, there may be several individual habitat banks where developers directly purchase credits. Here, I use the term to refer to a new kind of offsetting scheme where the habitat bank is an intermediary. It acts as a broker on the market, guaranteeing the quality and maturity of credits. The intermediary may keep mature credits of different habitat types in anticipation of future land use change. Landowners produce biodiversity offsets by conserving and restoring valuable habitats, and developers buy offsets to compensate for the biodiversity loss caused by development projects. Thus, a market for offsets emerges. Intermediaries in offset markets have been examined, for instance, in a case study by Coggan et al. (2013a). I analyse the role of an intermediary analytically. I develop an equilibrium model to study equilibria in offset markets: prices, potential size of market and realisation of risks associated to uncertainty.

I apply the analytical model to Finnish economy and three selected habitat types: abundant mires, scarce herb-rich forests, and expensive and laborious rural biotopes. The supply of offsets is estimated based on the data on areas suitable for habitat restoration, restoration measures and associated costs. I examine the evolvement of habitats under uncertainties by conducting a survey to specialists on each chosen habitat. To examine the impacts and risks of uncertainties, I perform Monte Carlo simulation. I estimate demand based on predictions of future land-use change: the increase of built-up areas and infrastructure. In addition to the role of the intermediary in the market, I also explore how the market equilibrium depends on trading ratios (rates of exchange). Trading ratios differ depending on whether biodiversity losses from development are ecologically equivalent to gains from compensation or not. The presence of the intermediary also affects the trading ratios as there is a time delay between losses and gains, which must be discounted to present time if the intermediary is not in the market guaranteeing mature offsets.

The thesis is structured as follows. In the next chapter, I introduce biodiversity offsetting as a conservation mechanism, as well as the current state of offsetting schemes around the world. I present the challenges in defining and implementing offsetting which is widely debated in the literature. Scientific literature on the economics of biodiversity offsets is also briefly introduced. In Chapter 3, I will present the equilibrium model analytically and also in parametric form, which allows us to apply the model empirically to Finnish data in Chapter 4. In Chapters 5 and 6, I present the results. First, I examine how the trading ratios impact market equilibrium and second, I add a fee and a risk premium collected by the intermediary and see how the market equilibrium changes. Finally, conclusions and discussion can be found in Chapter 7.

2. Biodiversity offsets: literature review

2.1 What are biodiversity offsets?

Biodiversity offsetting (also called ecological compensation) is a market-based mechanism for biodiversity conservation. Offsets are designed to compensate for unavoidable biodiversity loss caused by economic activity. The basic idea is simple: a developer must provide an improvement in biodiversity so that the lost ecological value is compensated. Offsetting allocates the costs of conservation on those responsible for habitat degradation, thus implementing a spoiler pays principle. Usually, the aim of biodiversity offsetting schemes is to achieve no net loss of biodiversity (NNL). Alternatively, net gain is a more ambitious objective adopted by some programs. The no net loss objective resembles an emission cap in the trading schemes as it sets a limit to biodiversity loss caused by development (McKenney & Kiesecker 2010; OECD 2016, 40.)

Commonly, offsetting is used as the final step of a so-called mitigation hierarchy: offsets are the last resort and compensate only for residual impacts after appropriate efforts have been made to avoid damage to ecosystems, to minimise all the unavoidable impacts and to restore biodiversity on-site (Bull et al. 2013; McKenney & Kiesecker 2010). The mitigation hierarchy is represented in Figure 1. The vertical axis represents loss of biodiversity. Avoiding, minimising and restoring impacts on-site reduces biodiversity loss. The residual loss, an orange box, can be compensated. Offsets cannot be used to reduce a developer's obligation to avoid, minimise and mitigate harm and they should not aim to ease a permit process – the mitigation hierarchy is intended to safeguard that offsets are not a license to trash (McKenney & Kiesecker 2010).

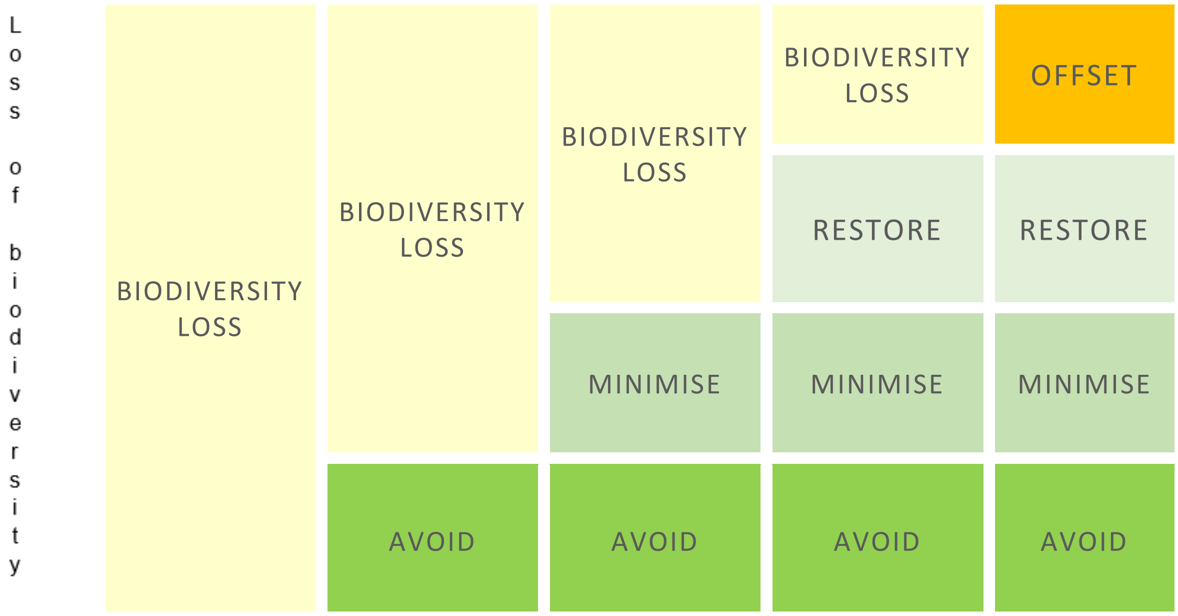

Figure 1. Mitigation hierarchy: developer must avoid and minimise damage, as well as restore biodiversity on-site. Offsets are used to compensate for the residual biodiversity loss. Source: Adapted from Rio Tinto (2012) "Rio Tinto and biodiversity: Working towards net positive impact".

All impacts cannot be compensated. Damaging extremely vulnerable ecosystems and habitats or endangered species should be avoided at all times. It is generally agreed that offsetting should be additional to existing biodiversity conservation efforts, beyond a counterfactual scenario (such as the amount of protected areas) (Gardner et al. 2013; Pilgrim and Ekstrom 2014). Consider Figure 2 where the state of biodiversity is degrading and existing conservation efforts increase over time to halt the degrading trend. Offsets are an independent addition in the policy mix, they do not affect the other conservation efforts since offsets cannot be used to compensate losses for protected areas or species. Ensuring that offset activities do not lead to the leakage of harmful activities and damage elsewhere is also essential (Gardner et al. 2013).

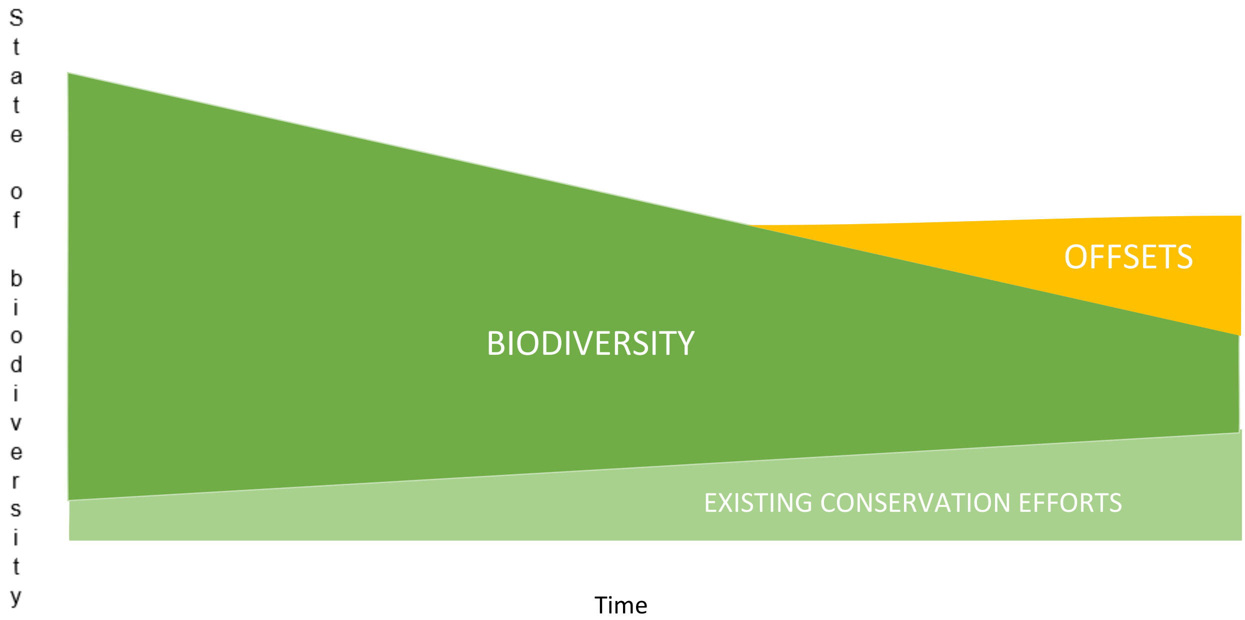

Figure 2. Offsetting should be additional to existing biodiversity conservation efforts. Source: Olli Ojala

Offsetting schemes provide several incentives to conserve biodiversity (Calvet et al. 2015). As offsets cause significant costs for developers, they will reduce impacts on biodiversity by minimising land use and allocating the development in lands with less valuable habitats. Also, developers will fill their offsetting requirement in the most cost-effective manner, that is, they will choose the most effective conservation projects with least costs. Landowners have an incentive to invest in the production of offsets which enables large and expensive restoration projects.

Offsetting can be implemented by legislation or it can be based on voluntary action (OECD 2016, 66-74). Economic agents can benefit from voluntary offsetting, as they can derive economic advantages, gain the support of local communities and NGOs, manage reputational risks or enhance corporate responsibility. The offsetting requirement can be carried out by restoring degraded habitats, creating new habitats and in some cases, preserving existing valuable ecosystems. There are currently three offsetting mechanisms. (OECD 2016, 50-53).

Direct offsets (one-off offsets, permittee-responsible mitigation) require developers to carry out compensatory measures themselves, case-by-case. It offers flexibility to address project-specific impacts but consistency, transparency and spatial coherence may be inadequate. Temporal loss of biodiversity is often unavoidable. Voluntary offsets are typically implemented in this manner. Often one-off offsets are the first step towards more developed compensation mechanisms. Offsetting funds (in-lieu-fees) are collected from developers to carry out restoration actions or conservation projects. Temporal loss of biodiversity is inevitable and additivity is not always carried out. Also, compensations do not fully correspond to the losses. (Calvet et al. 2015; OECD 2016, 50-53.)

The third mechanism to implement offsetting, habitat banking, is studied in the thesis. Habitat banking entails a third party implementing larger restoration projects ahead of future impacts, providing offset credits for developers to purchase. Thus, a market for offsets emerges and depending on the design, the risk of temporal loss may be removed. I analyse the market under different design scenarios, which affect whether temporal loss exists. With habitat banking, spatial context is more carefully considered and offsets are usually larger in size. (Briggs et al. 2009; OECD 2016, 52.) Sufficient trading activity and market size, i.e. adequate supply and demand for offset credits, are essential in creating a well-functioning market (Wissel & Wätzold 2010). In order to create sufficient demand, legislation on offsetting might be needed. Voluntary offsets likely lead to a thinner market. Supply depends on demand, as well as the existence of suitable areas for restoration, landowners' interests and the expected profits of producing offsets.

2.2 Offsetting schemes around the world

In 2011, there were 45 existing offsetting programs around the world, ranging from active habitat banking to programs channelling in-lieu-fees, to one-off offset policies. Furthermore, there were 27 programs in various stages of development or investigation. At least 187 000 hectares of land is under conservation management each year. There is a lack of comprehensive data on the different national offset schemes, but the turnover of compliance-based and voluntary biodiversity offset programmes is estimated to be more than USD 3 billion per year, growing at an annual rate of 10 %. (Madsen et al. 2011; OECD 2016, 23). In this section, I present a few of the offsetting schemes in greater detail.

The United States has the most mature and largest biodiversity markets. Wetland and stream mitigation programme has been established in the 1970s, and it aims to offset residual impacts to wetland functions and values. Compensatory mitigation can be carried out with mitigation banks, in-lieu fee programmes and permittee-responsible mitigation (one-off offsets). Mitigation banks are the preferred compensation option, and in-lieu-fees are preferred over one-off offsets. Mitigation banks are usually operated by private entrepreneurs who seek to make a profit. Single-user banks also exist, run by state agencies or private companies that regularly need compensation. In addition, there are a few non-profit mitigation banks. The number of available credits is established in the bank approval, and developers can purchase credits if regulators approve that the mitigation bank represents appropriate compensation. The bank owner is responsible for the success of the compensation sites, and liable in the case of failure. Approximately 450 000 acres have been permanently protected in wetland and stream banks in the US, roughly 22 000 acres each year (OECD 2016, 132-166).

Conservation banking, established in the 1990s, is another offsetting scheme in the US. It has an objective to offset adverse impacts to species in accordance with Endangered Species Act, rather than replace wetland functions and values. Similar to wetland mitigation, offsetting can be carried out with mitigation banks, in-lieu fee programmes and permittee-responsible mitigation. California is the largest participant in conservation banking in the US. In year 2011, approximately 74 800 acres had been permanently protected under conservation banks. On average, 4 400 acres were added to the program annually over the last 10 years. (Madsen et al. 2011; McKenney & Kiesecker 2010.)

In Europe, compensation mechanisms continue to gain recognition as a policy tool. The Netherlands, France, Switzerland, the United Kingdom, Germany and Sweden have developed or are developing offsetting schemes (OECD 2016, 29-36). In the member states of the EU, compensation is required if a development project affects the Natura 2000 network. It is a network of rare habitat types and core breeding and resting sites for rare and threatened species, and it aims to protect natural diversity in accordance with the EU's Birds Directive and Habitats Directive. The goal of compensatory measures is not to achieve no net loss but to maintain overall ecological coherence of the sites. Compensatory measures can consist of restoring or re-creating the same habitat as degraded, or in exceptional cases proposing a new site. Offsets for birds must be along the same migration path and "accessible with certainty by the birds usually occurring on the site affected by the project". (European Commission 2000; McKenney & Kiesecker 2010.)

In Germany, compensation for development-related biodiversity loss has been required since the 1970s. Compensation was originally carried out by the developers and because of strong functional, spatial and temporal requirements, there was little flexibility in the selection of possible sites, and consequently compensation projects were small scaled, highly fragmented and very costly. Requirements were relaxed in the 1990s with reforms aimed to improve ecological effectiveness and to make it easier and less costly to find appropriate compensation sites. Third parties could offer compensation measures and the concepts of land pool and eco-account were also introduced. Land pools are sites that are held aside for future compensation measures. An eco-account is a registry in which compensation measures that may be used in the future to compensate biodiversity losses are recorded. Losses and compensations are measured with a grading system, and the measurement unit is an eco-point. (OECD 2016, 176-192.)

In Hessen, Germany, there is a special intermediary, an eco-agency. Established in January 2006, it was the first legally recognised intermediary agency in Germany. It sets up land pools for areas usable for compensation and carries out compensation measures to provide eco-points so that developers can directly compensate their impacts. Also, it acts as an intermediary agent between eco-point suppliers and developers. It also helps to secure continuous management measures (for 30 years) if needed. Legally, the developer has the liability to provide these management measures but if the developer buys eco-points from the agency, the agency takes over the liability of the continuous management measures. The tasks of the eco-agency are carried out by a non-profit company, and majority of its shareholders come from the public sector. Public actors are also important suppliers of eco-points because they need to ensure that sufficient eco-points are always available for compensation so that economic development is not hindered. (OECD 2016, 176-192.)

In France, the mitigation hierarchy was incorporated into environmental law in 1976 but offsets were mostly ignored until the 2000s. In 2012, the French government published guidance on the mitigation hierarchy. It outlines that the aim of the mitigation hierarchy is to achieve no net loss of biodiversity, and preferably a net gain for currently threatened biodiversity and ecosystems. Offsets are a key mechanism for achieving no net loss. Offsetting is required for impacts on forests, wetlands, and protected species, among others. Legal requirements were strengthened to monitor and more effectively implement measures aimed at avoiding, reducing and offsetting impacts. However, although there are strong legal and financial implications for developers, the guidance does not consider the required institutional arrangements and science base. Local and regional permitting authorities and developers must plan and build adequate institutional arrangements. Consequently, demand for offsets has increased but the results are highly variable and ineffective. (Quétier et al. 2014.)

In Australia, offset policies have developed rapidly. There are offset schemes in New South Wales, Victoria, Western Australia, South Australia, and Queensland. Mostly the focus is on offsetting impacts to native vegetation. New South Wales has introduced BioBanking, a market-based approach involving ecosystem and species credits. Brazil, Mexico and India have mechanisms to offset impacts on forests, and Canada has established a scheme to compensate for losses to fish habitats. Also, countries including South Africa, China and Colombia have national offset policies. (McKenney & Kiesecker 2010; OECD, 29-36.)

2.3 Measuring and matching biodiversity: how to achieve no net loss

While more conservation outcomes may be achieved with biodiversity offsets, they are not a panacea for halting biodiversity loss. Ecological risks as well as the theoretical and practical challenges of offsetting are widely discussed in literature. These problems include the difficulties of measuring biodiversity, matching losses and gains, time lags, uncertainty, perverse incentives, non-compliance and lack of monitoring (e.g. Bull et al. 2013; Gardner et al. 2013; Gordon et al. 2015; McKenney & Kiesecker 2010; Maron et al. 2012; Maron et al. 2016; Quétier et al. 2014; Quétier & Lavorel 2011).

Bull et al. (2013) consider several conceptual challenges associated with biodiversity offsets. Biodiversity has no universal, unambiguous definition but offsets rely upon an accurate quantification of losses and gains, and to fully compensate damage to biodiversity, robust metrics is required. Instead of single metrics, compound or multiple metrics, which summarize ecological information about the site, is preferable. Also, the baseline against which to measure the loss of biodiversity is rarely specified. In dynamic ecosystems, no net loss should be defined against prevailing trends in biodiversity condition and take into account the business-as-usual scenario (Maron et al. 2015). The possible leakage of development impacts outside the area evaluated should be considered as it also has an effect on whether the objective of no net loss is really achieved. In addition, an important question regarding quantification is whether offsets should provide compensation for biodiversity, ecosystem function, ecosystem services or all three.

The ecological equivalence and matching of biodiversity losses and gains are also a subject of discussion in literature (Bull et al. 2015; Quétier & Lavorel 2011). As every ecosystem is unique and there are no ecologically identical habitats, it is difficult to compensate degradation with ecologically equivalent biodiversity components. Sites differ in type, location, time and ecological context – even when trading in kind (i.e., like-for-like). In-kind offsets provide habitats, functions, values, or other attributes similar to those degraded. Typically, trading out of kind is not preferable. However, by allowing out-of-kind offsets, trading up becomes possible (trading losses in a habitat of low conservation significance for gains in more valuable habitats). Guidance also differs on how proximate offsets need to be to an impacted site. The geographical area can be defined by catchment area, soil type or the location of affected species and populations. (McKenney & Kiesecker 2010.)

Determining how long offsets are expected to last is another important subject of debate. Permanent offsets may be required if project impacts are assumed to be irreversible, whereas offsets with fixed term may be allowed if there is potential to reverse damage at the project site. However, it is challenging to determine what is ‘permanent’ and what the management implications will be. It is not always clear how an offset should be maintained, by whom, and for how long. In addition, non-compliance with offset requirements and insufficient monitoring by regulators have been significant challenges in existing schemes. (Briggs et al. 2009; Bull et al. 2013; McKenney & Kiesecker 2010.)

While the damage caused by development for species and habitats is certain, the conservation outcomes of offsetting are not. This is often accounted by increasing the amount of compensation required. Trading ratios (multipliers, offset ratios) have been examined in several studies (e.g. Bull et al. 2016; Gibbons et al. 2015; Laitila et al. 2014; Moilanen et al. 2009). A trading ratio defines the rate of exchange between the loss of biodiversity and the gain achieved with compensation. They can be determined based on uncertainty, the ecological equivalence of loss and gain, and chosen compensation mechanism (McKenney & Kiesecker 2010). Time discounting is included in the trading ratio if immediate loss is compensated by uncertain future gains. Temporal loss of biodiversity is problematic because they may have detrimental impacts upon the wider ecosystem, or they may cause a temporary lack of ecosystem service provision (Bull et al. 2013). Time delays may be removed by implementing a banking mechanism, depending on the design (Bekessy et al. 2010).

Moilanen et al. (2009) calculated sufficient trading ratios to achieve no net loss and included the effects of uncertainty, correlation, and time discounting. Uncertainty and allowing for the possibility that restoration fails completely will increase the trading ratio substantially, and the influence of time discounting can be large as well. However, correlation in restoration success between different areas had the greatest influence on the ratio. They conclude that the trading ratio increases quickly from two to hundreds when such assumptions are accounted for.

Bull et al. (2016) collated information on trading ratios used in practice and conclude that the majority of proposed ratios are above 1.0 but less than 10.0. Realized ratios are generally at the lower end of the range proposed by policy. Their analysis of trading ratio values through time showed no significant increase. However, the substantial growth in scientific literature on offset design since the early 2000s is reflected in more detailed trading ratio requirements under recent offset policies and it is likely that trading ratio values will increase in the future as research findings are taken into account.

2.4 Economic studies on offsetting

Economic analysis on biodiversity offsetting schemes is limited to few. Most of the studies focus on the production and implementation of offsetting, dealing with the release of offset credits, incentives, investments and costs (Fernandez & Karp 1998; Bonds & Pompe 2003; BenDor et al. 2014; Coggan et al. 2013a; Coggan et al. 2013b; Hartig & Drechsler 2009). While providing understanding on many issues impacting market outcomes, these studies do not provide analysis at a market level, only some specific feature of the market. Here, I briefly present the findings of the economic studies on offsetting.

Fernandez and Karp (1998) develop a model to determine how much investment in habitat restoration is optimal, i.e. at what state of wetlands quality is it optimal to stop the restoration and "cash in". They conduct sensitivity analysis of how restoration costs, stochastic biological growth, an interest rate, and the price of credits affect the optimal level of investment. The study reveals that the highest level of restoration occurs when the costs of restoration decrease, biological uncertainty increases and the value of wetland credits increases. Bonds and Pompe (2003) model the cost minimization conditions for wetland mitigation banking, and included trading ratios and location in the credit calculation. They show that including location in the trading ratio in order to adjust varying levels of productivity at different locations will increase wetland conservation values but will not complicate credit calculation or affect administrative costs.

Coggan et al. (2013a) study three cases from Australian offset markets to assess if and how intermediaries reduce transaction costs to offset buyers and sellers. They identify six types of intermediaries: information providers, brokers, offset aggregators, banks, in-lieu-fee intermediaries and clearing houses. They find that transaction costs, generated by asset specificity, uncertainty and transaction frequency, are lowered by the intermediaries. The reasons behind the lowering costs are provision of information, as well as time and information intensive services, such as negotiation, monitoring and reporting. Because of specialisation in these areas, the intermediaries can most likely provide these services at a lower cost than the buyers and sellers would face if they performed the tasks themselves. They also find that the presence of the intermediaries cannot be explained by the ability to reduce transaction costs due to probity hazards; offset transactions that generate adverse ecological outcomes. Public intermediaries are not operating to reduce probity hazards from private intermediaries, and intermediaries cannot even generate probity hazards because the market is strictly regulated by a policy administrator.

In another study, Coggan et al. (2013b) conduct an analysis of the factors that influence transaction costs and how the influence occurs, with two Australian offset schemes as case studies. Theory identifies four influencing factors: transaction and transactor characteristics, institutional environment and institutional arrangements. They find that all four categories have an influence on transaction costs. The degree of influence and the importance of each factor varies across the two policies and between the parties in the market. Asset specificity and policy design have a particularly notable impact. Because of the specific ecological nature and the objective of offsetting, knowledge gained from past experiences is usually not transferable to new offset transactions. High asset specificity also makes it costly to generate consistent, generally relevant rules for all trades, which influences transaction costs through uncertainty and is particularly influential to the transaction costs of regulators and offset buyers. From a policy perspective, clear rules have a direct impact on buyer and regulator transaction costs. Indirectly, policy design has also a moderating effect on other influencing factors.

Hartig and Drechsler (2009) use an ecological-economic simulation model to examine how spatial connectivity may be considered in the financial incentives created by a market for offset credits. They simulate land use decisions with an agent-based model of land users. In addition, a metapopulation model evaluates the conservational success of the market. They find that offset markets that consider connectivity lead to considerably better conservation results than markets without spatial incentives. Optimal spatial incentives depend on species characteristics, such as dispersal distance, but also on the spatio-temporal distribution of conservation costs. Including spatial incentives may improve the efficiency of the offset markets considerably, especially when fragmentation is a significant threat for the impacted populations.

BenDor et al. (2014) explore the optimal degree of advanced credit release in biodiversity offset markets. The advanced credit release policy means that offset suppliers receive an upfront payment before it is verified that restoration actually succeeds, and another payment for the remaining credits if the restoration project proves to be successful. There is a tension between regulators' desire to induce market participation, while ensuring that offset suppliers complete the restoration after the advanced credit release. They conclude that regulators should select release rates such that more credit producers will participate in the market but selecting the zero effort of restoration is not incentivized. They show that as advanced release rates rise, the restoration effort decreases and profits increase.

Unlike others, Doyle and Yates (2010) examine offset markets analytically and empirically. They analyse interactions between economics and ecology for offset markets under two alternatives to no net loss regulation, NNL of ecosystem size and NNL of ecosystem function. An economic model of free-entry equilibria is linked with an ecological model. They include returns to scale and inefficiency of restored ecosystems to the model and apply their approach to wetland mitigation banking in North Carolina. In accordance with economic theory, a market modelled using free-entry equilibria is characterised by excess entry: the equilibrium number of firms is greater than the welfare maximizing number. Thus, they consider the effect of market entry on the number and size of restoration projects and examine, whether ecological considerations exacerbate or ameliorate excess entry.

Doyle and Yates (2010) find that in order to achieve NNL, both economic and ecological processes must be accounted for. Market entry, the efficiency of restored ecosystems, and the relationship between the ecosystem function and size of the restoration project (returns to scale) affect the choice of a trading ratio. The trading ratio is highly sensitive to returns to scale. When the efficiency of a restored ecosystem decreases, the trading ratio must increase. Also, when returns to scale increase, the trading ratio decreases. When ecological factors are considered, both excess entry and insufficient entry may occur on the offset market, regardless of the objective being NNL of ecosystem size or NNL of ecosystem function.

There is a wide and expanding scientific literature on biodiversity offsets. However, the emphasis of the literature is on the ecological consequences, theoretical aspects, and challenges in designing and implementing offsetting, and in comparison, economic studies are small in number and mostly focus on some specific features of the market. The previous economic literature shows that the design of institutional settings may affect a lot of market outcomes: equilibrium prices, the potential size of the market, transaction costs and risks associated to uncertainty. My objective is to increase the understanding of the economic basis of offsets on the market level. In the application of the model, I take into account the current science base. To define the baseline against which to measure the loss of biodiversity in the selected habitat types, I designed an expert survey, and I consider how the ecological equivalence and matching of biodiversity losses and gains, trading ratios and intermediaries affect market outcomes. I use compound metrics and the possible leakage of development impacts outside development sites are considered.

3. Economic Model

To analyse the biodiversity offset markets, I develop an equilibrium model. I examine the role of an intermediary, providing information and broker services. The intermediary guarantees that investments in restoration have been made before the need for offsets emerges and safeguards against risks if restoration do not succeed as expected. I take the supply of offsets to come from habitat restoration and nature management. Trading is based on a no net loss principle. The amount of credits needed to compensate for biodiversity losses is determined by using a trading ratio. The trading ratio is adjusted to match the ecological values of losses and gains.

The commodity traded in the market is an offset credit, which is calculated in terms of hectares and characterized by an ecological index value. It represents an increase in the ecological state of a restored habitat. The ecological state is measured by an index value, which varies between a natural state (1) and a completely degraded state (0). Consider Figure 3 where the ecological state of the habitat is on the vertical axis and the increase is measured as a difference between the ecological state of the habitat after restoration and management (an orange curve), and the state in a business-as-usual scenario (a blue curve), per hectare. The restoration effort immediately improves the state of the habitat and initiates the recovery towards the natural state.

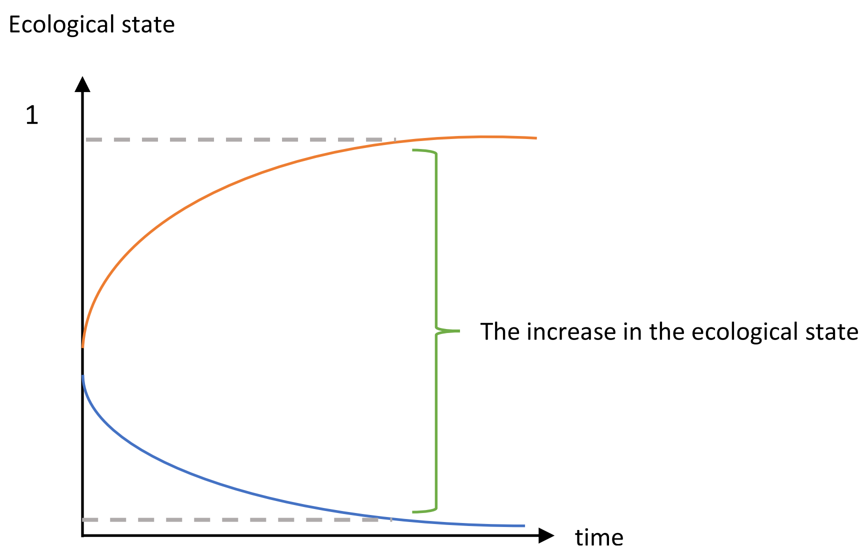

Figure 3. An offset credit represents an increase in the ecological value in the restored land area

3.1 Biodiversity offset market model

As a starting point, I consider a representative landowner making a habitat restoration investment in his/her land. This landowner takes an investment effort, x, to improve the habitat; the intensity of the effort may vary. The investment effort immediately improves the state of the habitat, which turns to an improving path in time, as seen in Figure 3. To formalize this idea, let an ecological function $f(\tau)$ represent the evolvement of the state of the restored habitat over time, and let a restoration function $A(x)$ describe the effect of the restoration investment on the evolvement of the habitat. τ represents a point in time when the improvement in the state of the habitat is measured, which is not generally fixed but based on an agreement. Thus, offset credits per parcel (q) are defined by $q = A(x)f(\tau) - \bar{f}(\tau)$, where $\bar{f}(\tau)$ is the state of the habitat in the business-as-usual scenario. Effort is costly, however. Habitat restoration entails a lump sum investment cost (F) and a unit cost related to effort (w). Let p be the price of offsets. The landowner is risk neutral, that is, uses the expected values, and maximises profits from habitat restoration in a given land area,

$$max\ \pi = p\left[A(x)f(\tau) - \bar{f}(\tau)\right] - wx - F(j), \tag{1}$$

where j is distance and $F'(j) > 0$ indicating that fixed costs increase in distance.

The choice of the effort is implicitly determined by

$$\pi_x = pA'(x)f(\tau) - w = 0. \tag{2}$$

By equation 2, the optimal effort is chosen by equating the marginal revenue from restoration to the unit cost of the effort. Solving for the effort gives: $x = [pf(\tau) - w]/A'(x)^{-1}$, where $^{-1}$ marks inverse function. Thus, the choice of effort depends positively on the price of offsets and negatively on the unit price of effort. Equation (2) holds for any land parcel. How many parcels does the landowner restore? Assume that moving to remoter areas increases the costs. Let π^A denote the return to land in an alternative use. There is a distance, which defines the last land parcel restored.

This distance is defined by: $j^*: \pi(x^*) = \pi^A$. This condition together with equation (2) defines the supply of offsets as a function of offset prices and costs: $q^s = \int_0^{j^*} A\big(x^*(p,w)\big) f(\tau) g(j) dj$. Offset supply is an increasing function of offset price,

$$q_p^s = \int_0^{j^*} A'\big(x^*(p,w)\big) f(\tau) \frac{\partial x}{\partial p} dj + A f(\tau) \frac{\partial j}{\partial p} > 0. \quad (3)$$

Next I turn to the need for offsets. A representative developer building, for instance, a production facility, or developing an area for utilization (a mine for instance) causes biodiversity loss and needs to buy offsets for compensation. How much the developer needs compensations, depends on the profitability of the development project and the extent of the loss. Thus, offsets provide the developer utility by facilitating the profitable business. Following Doyle & Yates (2010), the developer maximises his quadratic utility from offsets over costs

$$max\, U(q) = aq^d - \frac{b}{2} {q^d}^2 - pq^d, \quad (4)$$

where a and b are positive constants, q^d is the biodiversity loss requiring offsets matching this loss, and p is the price of offset credits. Choosing q^d yields $a - bq^d - p = 0$. The marginal utility derived from offsets equals the offset price. This condition gives the demand function, $q^d = (a-p)/b$, which is downward-sloping: $q_p^d = -\frac{1}{b} < 0$.

The market equilibrium can now be defined based on the choices of the representative supplier and demander. Lastly, the trading ratio is imposed, denoted by σ. If the amount of biodiversity loss is ecologically equivalent to gains, for like-for-like compensations, the trading ratio is equal to one. For like-for-better compensations or when the gain is higher than the loss, the trading ratio is less than one. When the loss is higher than the gain, or if trading for less valuable habitats is allowed, the ratio is greater than one. In the market equilibrium, demand for restored habitats depends on the required trading ratio, so that the equilibrium price is defined where demand meets supply:

$$p^*: q^d(\sigma) = q^s. \quad (5)$$

To illustrate how the trading ratios impact the market equilibrium, consider Figure 4. In order to fully compensate for habitat degradation, the trading ratio is adjusted to match the ecological values of biodiversity loss and gain on a case-by-case basis. In Figure 4, drawn are the downward-sloping demand function and the upward-sloping supply function. The equilibrium price is determined by their intersection and marked by $p*$ in both panels of Figure 4. Now, if the trading ratio exceeds unity ($\sigma > 1$), the demand curve moves outwards. The developer would be willing to buy the increased amount $Q**$ at price p' but must pay the new equilibrium price $p**$. If the compensation site is of better quality than the development site, the appropriate trading ratio is less than one ($\sigma < 1$), and the demand curve moves inwards. The developer will then buy decreased amount $Q**$ compensation at lower price $p**$.

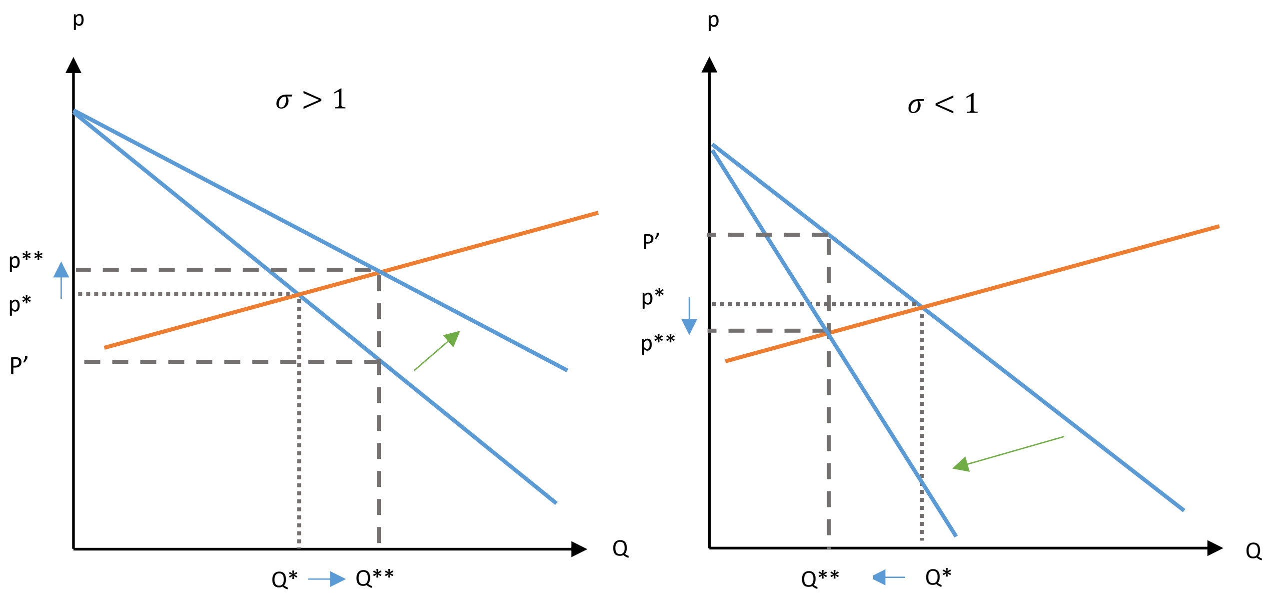

Figure 4. The impact of a trading ratio to the market equilibrium

3.2 Parametric analysis: scrutinizing supply

The previous analysis was general and in this section, I express the model in parametric forms to facilitate the scrutiny of alternative ecologically relevant cases of restoration. This leads to multiple details in the basic model. In the next chapter, I apply this parametric model empirically to Finnish data.

I use a sigmoid curve to model the evolvement of the habitat,

$$f(\tau) = \frac{L}{1 - e^{-k(\tau - l_0)}}. \tag{6}$$

L sets the maximum point of the curve, i.e. the natural state of the habitat, and k determines the slope of the curve. The sigmoid curve provides many advantages for the analysis. Especially, it allows setting the central point l_0, which changes the starting point of the curve and thus, fixes the state of restored the habitat in the beginning, as restoration effort immediately improves the state of the habitat. The sigmoid curve is first strongly increasing (convex) and then decreasing (concave). It describes the evolvement of the habitat better than a linear approximation. It may be misleading, though, for cases where rare species emerge in the compensation area over time, making the curve convex later than sooner, but presumably these cases are rare.

Restoration accelerates the recovery of the degraded habitat towards its natural state. The effect of the restoration investment is added to equation (4) by using a multiplicative formulation as follows:

$$A(x)f(\tau) = [(\alpha - \beta x)x]\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}}. \tag{7}$$

Parameter $\emptyset$ determines how degraded the state of the habitat is before restoration (in the numerator, $L = 1$, to set a limit to the evolvement to the natural state). I consider four illustrative cases, which all are relevant to practical restoration projects and the offset markets. They illustrate differences between selling now and in the future, as well as the types of restoration costs and their timing. Case 1 is the simplest: only an upfront investment is required and no other costs accrue after the first restoration investment. Offsets are assumed to be saleable at once. In case 2, I assume that costs and revenue accrue in different time periods. This case is relevant, for instance, for herb-rich forests. After the initial investment in nature management, they usually require regular follow-ups. Rural biotopes are a laborious and expensive investment as they require yearly management. In both habitats, offset credits can be sold after n years. In case 3, a percentage (θ) of offset credits can be sold in advance and the rest after n years. Advanced credit releases are an important factor affecting the supply side of offset markets as they aim to incentivize suppliers to entry the market (BenDor et al. 2014).

Case 1. Upfront investment and immediate sales

Under the above specification, the landowner maximises profits from restoration investment (F) according to equation (7):

$$\pi = p[(\alpha - \beta x)x]\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} - wx - F. \tag{8}$$

The first and second order conditions are

$$\pi_x = p(\alpha - 2\beta x)\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} - w = 0 \tag{9}$$

$$\pi_{xx} = -p2\beta\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} < 0 \tag{10}$$

The first order condition shows that the optimal effort is chosen by equating the marginal revenue from restoration to the unit cost of the effort. The optimal effort is defined by the following explicit solution:

$$x^* = \frac{\alpha}{2\beta} - \frac{w}{p2\beta\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}}}. \tag{11}$$

Equation (11) provides the simplest optimal restoration effort. It is characterized by the restoration technology parameters (the first term) and the evolvement of the habitat weighted by the cost-price ratio (the second term).

From now on, $B = p2\beta\emptyset(\frac{1}{1-e^{-k(\tau - l_0)}})$. Thus, equation (11) is defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{w}{B}. \tag{11'}$$

Comparative statics reveals that the optimal restoration effort depends on the technology parameters α and β, the price, and variable costs as follows:

$$\frac{\partial x^*}{\partial \alpha} > 0; \frac{\partial x^*}{\partial \beta} < 0; \frac{\partial x^*}{\partial p} > 0; \frac{\partial x^*}{\partial w} < 0$$

In the production function, the technology parameters have opposite impacts: when parameter α in the numerator increases, the optimal effort increases, and when parameter β in the denominator increases, the optimal effort decreases. Also, the choice of effort depends positively on the price of offsets and negatively on the unit price of effort. An increase in the offset price leads to an increase in the optimal effort, whereas increasing costs decreases the optimal effort.

The optimal effort depends on the point in time τ and ecological parameters as follows:

$$\frac{\partial x^*}{\partial \emptyset} > 0; \frac{\partial x^*}{\partial \tau} > 0; \frac{\partial x^*}{\partial l_0} < 0; \frac{\partial x^*}{\partial k} > 0$$

The choice of effort depends positively on $\emptyset$, τ and *k*. Thus, the better the initial state of the habitat, the higher optimal effort is. Also, when the increase in ecological value is measured at a later point in time, the optimal effort increases. An increase in the slope of the curve increases the optimal effort as well. Parameter l_0 has a negative impact on the optimal effort. It fixes the starting point of the curve, as restoration effort immediately improves the state of the habitat. Thus, the smaller this impact is, the higher the optimal effort is.

Case 2. Costs and revenue accrue in different time periods

In case 2, costs and revenue accrue in different time periods. The revenue landowner receives is obtained at a future point of time, denoted by *n*. Furthermore, after the initial investment (F), additional restoration effort is needed and it takes place at time *m*. Thus, the profit function can be expressed as follows:

$$\pi = [p(\alpha - \beta x)x](1+r)^{-n} * \emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} - wx(1+r)^{-m} - F. \tag{12}$$

First and second order conditions are

$$\pi_x = [p\alpha - p2\beta x](1+r)^{-n}\emptyset\frac{1}{1-e^{-k(\tau-l_0)}} - w(1+r)^{-m} = 0 \tag{13}$$

$$\pi_{xx} = -p2\beta(1+r)^{-n}\emptyset\frac{1}{1-e^{-k(\tau-l_0)}} < 0 \tag{14}$$

The interpretation of the first order condition stays the same, only discounting is added. Again, the optimal effort is chosen by equating the marginal revenue from restoration to the unit cost of the restoration effort but the interest rate r in the discount factor impacts both the marginal revenue and unit cost of the effort. The optimal effort is defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{(1+r)^{-m}}{(1+r)^{-n}}\frac{w}{B}. \tag{15}$$

Comparing equation (15) with (11) reveals how discounting impacts. The difference lies in the term $(1+r)^{-m}/(1+r)^{-n}$. Clearly, the outcome depends on whether $m > n$ or not. In the former case, the size of the denominator $(1+r)^{-n}$ is decreased and effort is increased, and vice versa in the latter case. Economically, the further the costs of effort occur in the future, the higher the restoration effort.

Qualitatively, comparative statics stays the same. The impact of the interest rate depends on the relation between m and n. An increase in the interest rate increases the optimal effort if $m > n$ and vice versa if $m < n$.

$$\frac{\partial x^*}{\partial r} > 0 \; if \; m > n$$

$$\frac{\partial x^*}{\partial r} < 0 \; if \; m < n$$

Suppose instead that the restoration effort costs occur annually. I use annuity to discount the costs and the profit function is given by,

$$\pi = [p(\alpha - \beta x)x](1+r)^{-n}\emptyset \frac{1}{1-e^{-k(\tau - l_0)}} - wx\left(\frac{1-(1+r)^{-m}}{r}\right) - F \tag{16}$$

First and second order conditions are

$$\pi_x = [p\alpha - p2\beta x](1+r)^{-n}\emptyset \frac{1}{1-e^{-k(\tau - l_0)}} - w\left(\frac{1-(1+r)^{-m}}{r}\right) = 0 \tag{17}$$

$$\pi_{xx} = -p2\beta(1+r)^{-n}\emptyset \frac{1}{1-e^{-k(\tau - l_0)}} < 0 \tag{18}$$

Again, the interpretation of the first order condition remains the same, the optimal effort is chosen by equating the marginal revenue to the unit cost of the effort but now, as costs are discounted by using annuity, the unit cost of effort increases substantially. The optimal effort is now defined by:

$$x^* = \frac{\alpha}{2\beta} - \frac{[1-(1+r)^{-m}]}{r(1+r)^{-n}}\,\frac{w}{B}. \tag{19}$$

Relative to the previous case, the difference to the benchmark equation (11) is defined by the term $[1-(1+r)^{-m}]/r(1+r)^{-n}$. Irrespective of the size of n and m, $[1-(1+r)^{-m}]/r(1+r)^{-n} > (1+r)^{-m}/(1+r)^{-n}$. Thus, relative to the previous case, the latter negative term increases and when costs occur annually, optimal effort decreases. In this case, the effect of the discount rate is ambiguous. Otherwise, comparative statics remains the same.

Case 3. Right to sell a share of offsets immediately once the initial restoration investment has been made.

Again, restoration costs occur annually but now, a part of unrealized offsets can be sold immediately (advanced credit release), whereas the rest of offsets can be sold at a future point of time, n. Let θ denote the advanced credit release rate. Then, the profit function can be expressed as:

$$\pi = \theta[p(\alpha - \beta x)x]\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} + (1 - \theta)[p(\alpha - \beta x)x](1 + r)^{-n} * \emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} - wx\left(\frac{1 - (1 + r)^{-m}}{r}\right) - F. \quad (20)$$

First and second order conditions are

$$\pi_x = \theta(p\alpha - p2\beta x)\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} + (1 - \theta)[p\alpha - p2\beta x](1 + r)^{-n}\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} - w\left(\frac{1 - (1 + r)^{-m}}{r}\right) = 0 \quad (21)$$

$$\pi_{xx} = -\theta p2\beta\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} + (1 - \theta)p2\beta(1 + r)^{-n} * \emptyset \frac{1}{1 - e^{-k(\tau - l_0)}} < 0 \quad (22)$$

In the first order condition, the first two terms represent marginal revenue from restoration: the first term is the marginal revenue from advanced credit release and the second is the discounted marginal revenue from selling the rest of the credits after n years. The last term represents the unit cost of the effort, discounted with annuity. The optimal effort is now defined by

$$x^* = \frac{\alpha}{2\beta} - \frac{[1 - (1 + r)^{-m}]}{r[\theta + (1 - \theta)(1 + r)^{-n}]} \frac{w}{B}. \quad (23)$$

Relative to the previous case, the optimal effort increases. As $\theta > 0$ by assumption, $[1 - (1 + r)^{-m}]/r(1 + r)^{-n} > [1 - (1 + r)^{-m}]/r[\theta + (1 - \theta)(1 + r)^{-n}]$. The latter term now decreases as advanced credit release is allowed, and optimal effort increases.

Again, the effect of the discount rate is ambiguous but otherwise, comparative statics remains the same. In addition, increasing the advanced credit release rate θ increases optimal effort:

$$\frac{\partial x^*}{\partial \theta} > 0$$

The formal analysis of investments in producing offsets and the market is now complete. Next, I apply the model to empirical cases. It allows us to assess the magnitudes of the chosen timing and cost structures.

4. Data

4.1 Habitats and restoration measures

I apply the model to Finnish data with three habitat types: pine mires, herb-rich forests and rural biotopes. The habitats are representative to the Finnish nature and highlight differences in restoration costs and timing of the investment. I utilize the results of the working group on improving the status of habitats in Finland (ELITE report) for the valuation of the ecological state of each habitat, habitat specific restoration and nature management measures and the cost estimations of the investments.

The valuation of the ecological state of each habitat is based on habitat-specific structural characteristics, which are weighted according to their importance for biodiversity. Following ELITE, the ecological state can range from 0 to 1, where 1 is equivalent to a habitat in its natural state, or in the case of rural biotopes and herb-rich forests, the target state of the habitat. Equation (24) is used to calculate the state of the habitat in its current state (Kotiaho et al. 2015, 35-37):

$$R = \prod_{n=1}^{N} \left(1 - L_n \left(1 - \frac{n_{curr}}{n_{ref}}\right)\right). \tag{24}$$

R is the current state of the habitat, N is the number of structural characteristics, L_n is the weight indicating the importance of each characteristic to biodiversity. The weight is a percentage by which the state of the habitat degrades if that factor is completely lost. n_{cur} and n_{ref} are the current state and the natural state of characteristic n.

4.1.1 Pine mires

As the first habitat type, I focus on the restoration of oligotrophic pine mires. In this case, pine mires have been drained but peat harvesting and forestry are unprofitable. A mire is geologically defined as an area with a peat layer of at least 30 cm, but the area covered by mires is much larger in biological terms. Mires can be classified based on ecological gradients, such as on hydrology, supplementary vs. inherent nutrient influence, acidity and trophic status and mire water level. Pine mires are usually nutrient poor, with a thick peat layer. Typical species are pine, cotton grass, arctic cloudberry, many dwarf shrubs and sphagnum-mosses. There has been some 4.7 million hectares of pine mires in Finland, of which 2.8 million hectares are drained. (Kotiaho et al. 2015, 123-126; Raunio et al. 2008b, 173-174.)

Peatland drainage for forestry and peat harvesting are the most important causes of threats for mire species. Mire ecosystems started to degrade when more intensive peatland drainage began in the 1950s. Peat harvesting for energy production became common in the 1970s. In addition, infrastructure construction and groundwater abstraction have deteriorated mires. Nowadays only about 40 % of the original area of mires is left undrained in Finland. Approximately half of all mire habitat types are classified as threatened. The most threatened mire types are spruce mires, rich fens, and groundwater-influenced mires. (Rassi et al. 2010, 68-75.)

Currently, peatland drainage in undrained areas has almost ceased, yet drainage continues to be a significant threat to mire species as the draining effect of earlier operations continues in many areas and deteriorates the natural state of mires. In addition, the local drainage of mires can have adverse effects on other, pristine mires that are hydrologically connected. There is also pressure for increasing the use of peat for energy production. If peat harvesting is not restricted to mires that have already lost their natural state, effects on mire species will grow considerably. (Rassi et al. 2010, 68-75.)

Hydrology is the most important factor affecting the state of mire ecosystems. Drainage and other land use disturb hydrology by blocking water flow to the mire or increasing the outflow of water. Especially, drainage lowers the water level on mires and increases the number of trees. Thus, mire species are replaced with species adapted to a dryer and shadier environment. The ecological state of pine mires is presented in Table 1. Drawing on equation (24), the current state is estimated based on tree stand and hydrology. Hydrology is estimated roughly on a percentage: 100 % represents hydrology in the natural state, and 0 % represents completely degraded hydrology where natural water flow is non-existent. Hydrology is given more weight due to its importance to the state of mire ecosystems. The current state of pine mires is on average 0,32. (Kotiaho et al. 2015, 123-155.)

Table 1. Structural characteristics of pine mires

	Tree stand	**Hydrology**	**State of the habitat**
Natural state	20 m^3/ha	100 %	1,0
Weight	0,1	0,95	
Current state on average	30 m^3/ha	30 %	0,32

4.1.2 Herb-rich forests

The second habitat type I consider is herb-rich forests. Herb-rich forests are defined based on the characteristics of vegetation, soil, moisture conditions and tree species composition, and usually support several tree species. Takeover by spruce is natural in the late states of succession. Typically, herb-rich forests grow in eutrophic, slightly sour soil. In Finland, almost half of threatened forest species and more than 40 % of all red-listed forest species live primarily in herb-rich forests. Various types of herb-rich forests are also among the most threatened forest habitat types. (Rassi et al. 2010, 56-61; Raunio et al. 2008b, 262.)

The amount of decaying wood is one of the most important factors affecting the diversity of forest species. Forest management activities, changes in the tree species composition of forests, the reduction of old-growth forests, and the decreasing number of large trees are also significant threats. The decreasing amount of decaying wood is associated with all of them. In recent years, the use of wood provided by forests has been intensified, for example, by shortening the felling cycle and collecting logging residue and stumps for biofuel, which will further decrease the amount of decaying wood remaining in forests. (Rassi et al. 2010, 61-67.)

Forestry and the lack of natural disturbance dynamics have reduced the variety of tree species in forest stands for a long time. Wildfires and storms would naturally renew forests. The following succession would vary diversely due to the differences in habitats, living and decaying tree stands and other vegetation, size of the area, and other aspects of the surrounding environment. If succession continued without interruptions, several tree generations with different tree species compositions would appear. As nowadays forestry mainly regulates succession, forests are even-aged, they lack the diversity of early succession forests, and old-growth forests are rare. (Raunio et al. 2008a, 113-114.)

I refer to a target state instead of a natural state, as with management, herb-rich forests are kept in certain phases of succession to prevent the natural proliferation of spruce, which has a negative impact on the diversity of herb-rich forest species. In Table 2 are presented three structural characteristics that indicate the degradation of herb-rich forests: the number of large trees (with a diameter of at least 40 cm), the amount of decaying wood, and the volume of broad-leaved trees. These factors are significant for the diversity of forest species and forest habitats. Large trees are significant for predator birds and epiphytes. Also, large trees produce important large decaying wood. Broad-leaved trees are especially important for biodiversity in herb-rich forests and thus, their volume is given the biggest weight. Currently, the state of herb-rich forests is on average 0,44. (Kotiaho et al. 2015, 100-116.)

Table 2. Structural characteristics of herb-rich forests

	Number of large trees	**Amount of decaying wood**	**Volume of broad-leaved trees**	**State of the habitat**
Target state	30	100 m^3/ha	100 m^3/ha	1,0
Weight	0,4	0,4	0,6	
Current state on average	10,1	7 m^3/ha	92 m^3/ha	0,44

4.1.3 Rural biotopes

Finally, I analyse rural biotopes that are open, semi-natural dry, mesic and moist grasslands resulting from grazing or mowing, as well as wooded pastures and meadows. All rural biotopes are classified as critically endangered or endangered in Finland. Their area has declined by more than 90 % since the 1940s, with their quality considerably deteriorated. More than 20 % of red-listed species have cultural habitats as their main habitat. Conservation of rural biotopes requires continuous management. Currently 30 000 hectares of rural biotopes are managed (mostly with the help of Finnish national agricultural aids) while the minimum target to ensure the long term survival of the most important species is 60 000 hectares. (Kemppainen & Lehtomaa 2009; Rassi et al. 2010, 108-109.)

The most significant threat to rural biotope species is the overgrowing of meadows and other open habitats. Open habitats have closed up due to changes in farming and pasturing practices: the number of small or intermediate size farms has declined, and in many areas traditional grazing and mowing has either ceased or decreased considerably. Also reforestation, fertilisation and the atmospheric fallout of nutrients degrade rural biotopes. (Rassi et al. 2010, 113-116.)

Again, I refer to a target state instead of a natural state, as rural biotopes are shaped and maintained by human activities and loose the biodiversity characteristics typical for these habitats without constant or repeated management. In Table 3 are presented the structural characteristics to estimate the ecological state of rural biotopes: vegetation, the openness of the habitat, and the history of soil cultivation. Vegetation refers to the condition of the field layer. Overgrowth, eutrophication and incorrect management disturbs the plant species typical for rural biotopes. Openness of the habitat refers to the fact that rural biotopes are typically open grasslands, pastures and meadows with diverse field layer and few trees. The increasing number of trees and shrubs reduces the typical openness of the habitat and replaces the species adapted to open ecosystems. History of soil cultivation affects the plant species composition in the habitat. Fertilisation and turning the habitat to agricultural use is harmful as soil cultivation alters the structure of the soil and thus affects the plant species. The habitat is in its target state when all factors are in 100 % condition. The current state of rural biotopes is on average 0,06. (Kotiaho et al. 2015, 159-166.) Thus, the state of rural biotopes is the weakest in comparison with two other habitat types considered in this study.

Table 3. Structural characteristics of rural biotopes

	Vegetation	Openness of the habitat	History of soil cultivation	State of the habitat
Target state	100 %	100 %	100 %	1,0
Weight	0,85	0,75	0,95	
Current state on average	10 %	20 %	60 %	0,06

4.2 Restoration and nature management measures

Tables 1-3 provide the estimates of the current state and the natural or target state of the selected habitats. Next, I ask how the state of each habitat could be improved by restoration and nature management to initiate recovery, and how the habitats will evolve in time.

Landowners supply biodiversity offsets for compensation by restoring and performing nature management measures and thus, produce additional biodiversity gains that would not otherwise take place. Restoration includes measures that initiate or accelerate the recovery of an ecosystem towards its original natural state (Kotiaho et al. 2015, 25). Nature management refers to measures aimed at keeping a habitat in certain phases of succession which are the most important for biodiversity (Similä & Junninen 2011, 13). There is a wide array of methods which aim to enhance ecosystem recovery, and next I describe them in more detail for each selected habitat type. As it is important to ensure the long-term existence of the offsets, compensation areas are established as permanent conservation areas or preserved in other legally binding fashion.

For pine mires, restoration measures consist of filling drains and removing tree stand to an amount consistent with a pine mire in a natural state. When drains are filled, water level is expected to reach its natural state and the flow of water recovers. Recovering hydrology is a precondition for the recovery of structure and functions of a mire ecosystem. Removing trees will restore the openness of the mire and partly affect water levels as well. (Kotiaho et al. 2015, 150-155.)

Herb-rich forests are restored using nature management measures that aim to create forests dominated by broad-leaved trees, with a diverse tree stand structure, decaying wood, and large trees. Although takeover by spruce is part of natural succession, it has a negative impact on the diversity of herb-rich forest species. Thus, the objective of nature management is not to achieve a forest in its natural state but to maximize biodiversity in the habitat. The nature management measures include reducing the number of spruces and managing the forest, regularly if needed, to prevent the natural proliferation of spruce, to increase the share of broad-leaved trees and to secure variation in the tree stand structure. Formation of decaying wood can also be promoted, where appropriate. (Kotiaho et al. 2015, 106-113.)

Rural biotopes require repeated management measures to maintain the preferred habitat conditions. If a rural biotope has been unattended for prolonged time, it requires a thorough renovation which includes thinnings, removing coppice and young trees, and mowing unfavourable vegetation. Thereafter the biotope is managed annually to prevent overgrowth and to maintain open areas. The repeated management measures are grazing and moving, which maintain the preferred habitat characteristics and enable the survival of fauna and flora typical for rural biotopes. Thus, the species composition characteristic of rural biotopes is maintained. The habitat is not cultivated, fertilised or managed with pesticides. Neither drainage nor reforestation are allowed. (Kotiaho et al. 2015, 159-166.)

4.3 The uncertainties of habitat recovery

There is some degree of uncertainty in how different habitats respond to restoration and management. To map the scope of these uncertainties, I designed and conducted an expert survey to examine the different scenarios of uncertainty. The objective was to estimate how the habitats would develop without restoration and/or nature management measures compared with a business-as-usual scenario, and how the outcomes of restoration and nature management measures vary under uncertainties. The survey was conducted for each habitat type separately and the respondents were experts specialized in the ecology of the habitat in question. Ten respondents received the surveys regarding pine mires and rural biotopes, and the survey for herb-rich forests was sent to nine respondents. The experts represent the up-to-date understanding of these habitat types in Finland.

In the introduction of the survey, both the habitat type and the specific restoration measures were described as in this thesis. I asked first the respondents to estimate based on their best knowledge how the habitat would develop without any restoration measures in a hundred years and in two hundred years. Also, the respondents were asked to give estimates for the most likely value as well as minimum and maximum values for the possible outcomes of habitat restoration after a hundred years. Different scenarios for restoration outcomes are presented in Figure 5. The respondents were also asked to estimate probabilities for each value. Finally, the respondents estimated a probability for the restored habitat to reach a natural state after two hundred years. In addition, after each question, the respondents were asked to give a confidence level for their answer on a Likert scale. For the entire survey, see Appendix C (in Finnish).

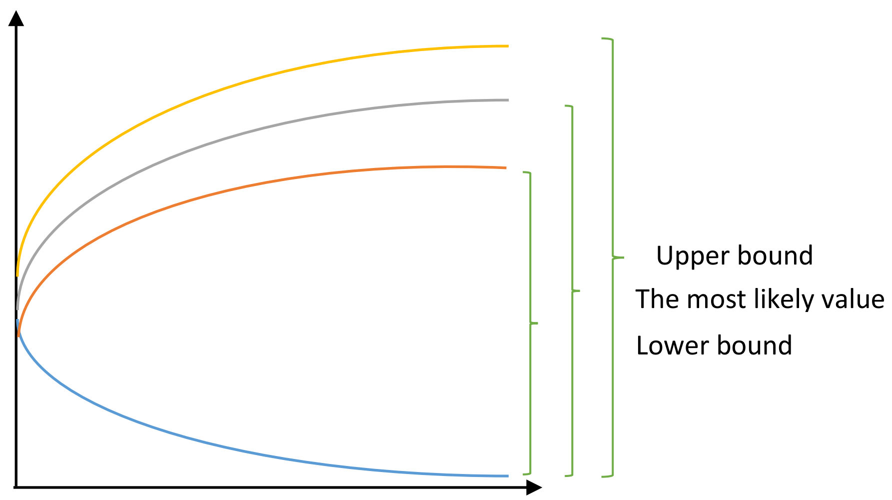

Figure 5. Lower and upper bounds and the most likely values for habitat restoration outcomes were asked in the survey. Blue curve represents scenario without any restoration or nature management measures, and yellow, grey and red curves represent three scenarios for restoration outcome.

I received seven responses on pine mires, six on rural biotopes and four on herb-rich forests. However, two responses for pine mires and rural biotopes had to be rejected as there were inconsistencies in the given probabilities. The responses are presented in Table 4 and Figure 6 below. The averages are weighted based on the given confidence levels.

Table 4. Weighted averages of survey responses

	Current state	Restored, 100 years			Business-as-usual	
		Lower bound	*Most likely*	*Upper bound*	*100 years*	*200 years*
Pine mires	0,32	0,63	0,79	0,89	0,45	0,6
Herb-rich forests	0,44	0,68	0,84	0,95	0,21	0,18
Rural biotopes	0,06	0,56	0,85	0,94	0,04	0,005

According to the responses, the state of pine mires develops towards natural state slightly without restoration (Table 4). Since it was here assumed that peat harvesting and forestry are unprofitable, drainage maintenance would stop, and hydrology would slowly start to recover. However, filling the drains and removing trees would cause the mire to recover faster towards its natural state. The state of herb-rich forests would decrease significantly without nature management measures, mostly due to forestry and the lack of natural disturbance dynamics. Natural management measures would restore the structural characteristics of herb-rich forests, above all by removing spruces and leaving broad-leaved trees standing. Currently, rural biotopes are in a highly degraded state and without management their state would gradually fall to zero. Thinnings, removing coppice and mowing unfavourable vegetation would restore the openness of the habitat, and annual management would cause the habitat to recover towards its natural state. The respondents were optimistic about succeeding in restoration: their estimate for the likelihood of the habitat reaching its natural state in 200 years was on average 92 % in pine mires, 94 % in herb-rich forests and 92 % in rural biotopes.

Table 5 reports the standard deviations of the answers. Despite the small sample, standard deviation is a helpful tool for observing the distribution of responses. The experts who participated in the survey were quite unanimous in their assessments, as variation in restoration results between individual answers in each survey was small. Variation between the habitats is also rather low, which means that there are not remarkable differences in the expected success of restoration of the habitats. The respondents were particularly unanimous regarding the evolvement of rural biotopes.

Table 5. Standard deviations of the responses show that the respondents were quite unanimous

	Restored, 100 years			**Business-as-usual**	
	Lower bound	*Most likely*	*Upper bound*	*100 years*	*200 years*
Pine mires	0,09	0,09	0,16	0,13	0,27
Herb-rich forests	0,10	0,06	0,18	0,06	0,10
Rural biotopes	0,08	0,06	0,08	0,05	0,01

I use equation (6) and Table 4 to illustrate the evolvement of habitats in Figure 6 over 200 years' time period. The upper lines represent the most likely case under habitat restoration based on the expert estimates, which show that habitats gradually approach their natural state. The lower graphs indicate how habitats were predicted to evolve over time if they were not restored or managed. They illustrate the above described finding that the state of herb-rich forests and rural biotopes would degrade over time and decline close to zero.

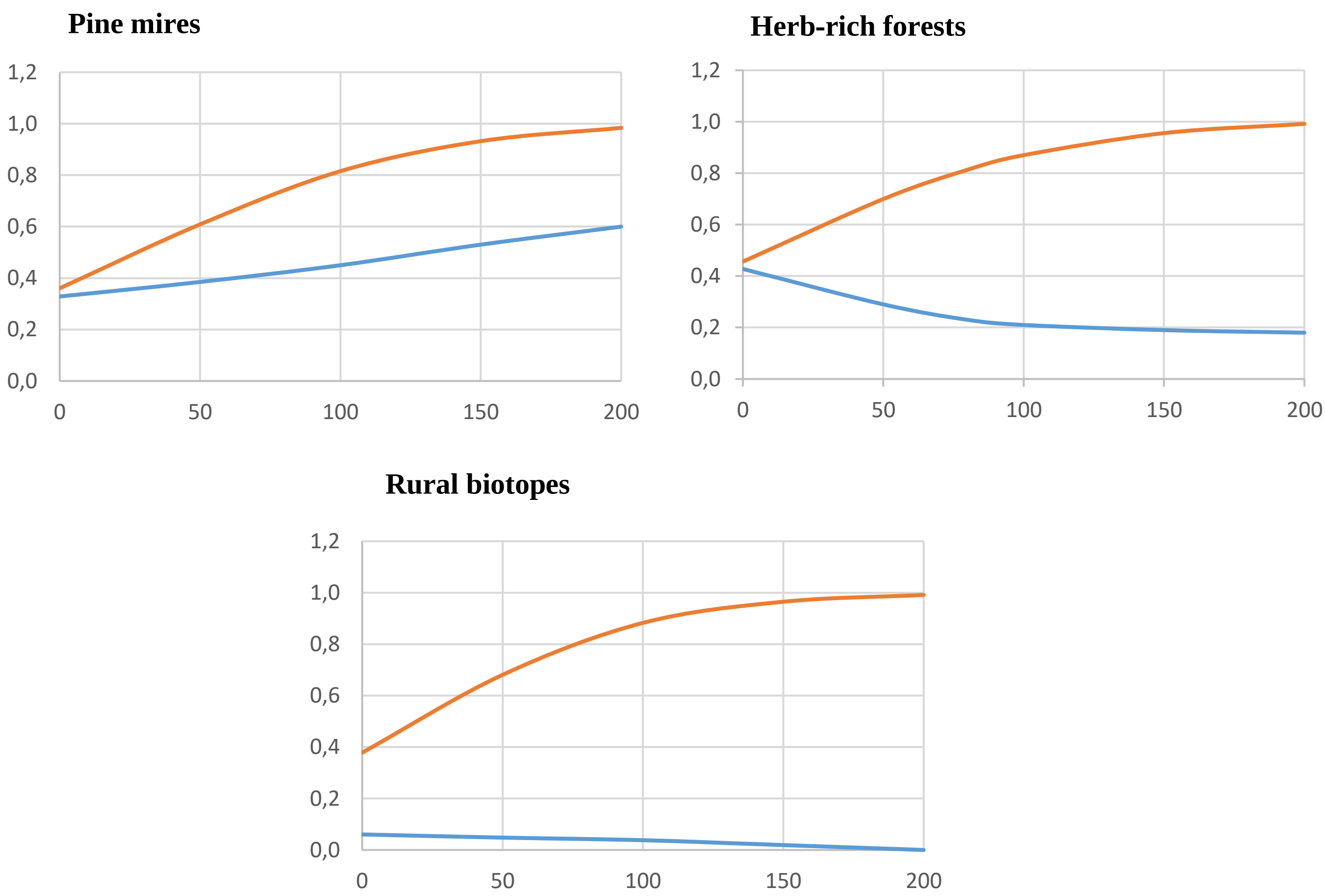

Figure 6. The evolvement of the ecological state of the habitats

Pine mires differed from herb-rich forests and rural biotopes as their state was estimated to be likely to develop towards the natural state also without restoration if they were left intact. Thus, the improvement in the ecological state, which would be achieved with restoration, was considerably lower in pine mires than in other habitats in this study. The increase is 0,34 after 100 years and 0,40 after 200 years whereas in rural biotopes the same figures are 0,81 and 1,0. As the traded offset credits are calculated by an ecological index value, representing an increase in the ecological state in a restored land area, this means that there are fewer offset credits for sale per hectare in pine mires than in other habitat types.

4.4 Parameters in the simulation model

In the previous sections, I introduced the habitat types selected for the application of the model, restoration measures and the evolvement of the habitats with and without restoration and management. Now, I represent the parameters in the model, which are scaled in accordance with data on the evolvement of the habitats, and costs of restoration and nature management. Recall equation (7) representing the evolvement of the habitat with restoration effort:

$$A(x)f(\tau) = [(\alpha - \beta x)x]\emptyset \frac{1}{1 - e^{-k(\tau - l_0)}}$$

Table 6 includes the parameters in the empirical application of the restoration model. Pine mires are an example of case 1, herb-rich forests represent case 2, and rural biotopes represent both case 2 with annually accruing costs, and case 3 with advanced credit release policy (see Appendix B).

Table 6. Parameters in equation (7)

		Pine mires	Herb-rich forests	Rural biotopes
technology parameter	**α**	3,9	3,9	3,9
technology parameter	**β**	2,1	2,1	2,1
degraded state of the habitat	**Φ**	0,62	0,565	0,77
central point of the curve	**l_0**	30	10	20
slope of the curve	**k**	0,02	0,02	0,025
point in time	**τ**	50	50	50

Technology parameters were scaled so that effort varies between 0-1 and full effort will cause the habitat to evolve optimally towards its natural or target state. The degraded state of the habitat is scaled so that it corresponds to the data represented in Tables 1-3. The central point and slope of the curve are scaled to correspond to the data on the evolvement of the restored and managed habitats, derived from the survey.

Table 7. Parameters in the profit functions

		Pine mires	Herb-rich forests	Rural biotopes
variable cost	**w**	1 400	150	918,75
fixed cost	**F**	1 000	9 264	6 849
discount rate	**r**	-	0,01	0,01
timing of credit sale	**n**	-	2	5
timing of variable costs	**m**	-	20 & 40	50

Now we turn to economic parameters. For cost estimates, I utilize data from ELITE report (Kotiaho et al. 2015). Some adjustments and corrections have been made to cost calculations concerning herb-rich forests and pine mires. The timing of an offset credit sale has been set at 2 years in herb-rich forests and 5 years in rural biotopes. Pine mire credits are saleable at once. The data on existing biodiversity markets, realized costs and offset prices is very limited. Thus, the parameters in the model have been scaled so that the market equilibrium is feasible and profits are in a benchmark case approximately 10 %.

Pine mires are usually restored once so there are only upfront costs. The price of timber is 30 €/m^3 in the calculations. A fixed cost, the total cost of conservation, is 1 000 €/ha and includes the value of the current tree stand (20 m^3) 600 €/ha and the value of land 100 €/ha. An administrative cost of 300 €/ha, related to establishing a conservation area, is also added. A variable cost, the restoration investment, is 1 400 €/ha. First, it includes the costs of removing trees (10 m^3), 1 000 €/ha, and 300 €/ha revenue from selling timber. Second, the cost of filling the drains with an excavator is 500 €/ha. A planning cost of 200 €/ha is also added. (Kotiaho et al. 2015, 138-140.)

Herb-rich forests require a larger fixed investment in nature management in the beginning and follow-ups (a variable cost of 150 €/ha) 20 and 40 years after the initial management investment (Kotiaho et al. 2015, 114). The nature management measures and their costs vary greatly between sites, depending on the state and age of the tree stand. Clearing a stand of spruce saplings can be extremely costly, whereas the removal of mature spruces can yield notable sale revenue. I updated the cost estimations presented in ELITE report to this thesis (for similar calculations, e.g. Ahtikoski et al. 2007).

First, the fixed cost of conservation is approximately 7 400 €/ha. I have calculated it as a bare land value for managed spruce forest land, with an added administrative cost of 20 % (Kotiaho et al. 2015, 114). Second, I have estimated the costs of nature management measures separately for a site dominated by spruces and for a site dominated by broad-leaved trees. I also took into account the age of the tree stand: costs are different for saplings, a young tree stand and a mature tree stand. For the cost of nature management measures, consider equation (25) and Figure 7. The vertical axis represents the growth in forest value and the horizontal axis represents time. Spruces are removed at time t'. The optimal rotation time is T^*. Thus, the landowner faces a cost for not clear cutting at time T^* but instead, removing only spruces at time t'. This cost is discounted by a factor $(1+r)^{T^*-t'}$. A planning cost of 150 €/ha is also added.

$$F = p\left[h(T^*)(1+r)^{T^*-t'} - h(t')\right] + 150 \tag{25}$$

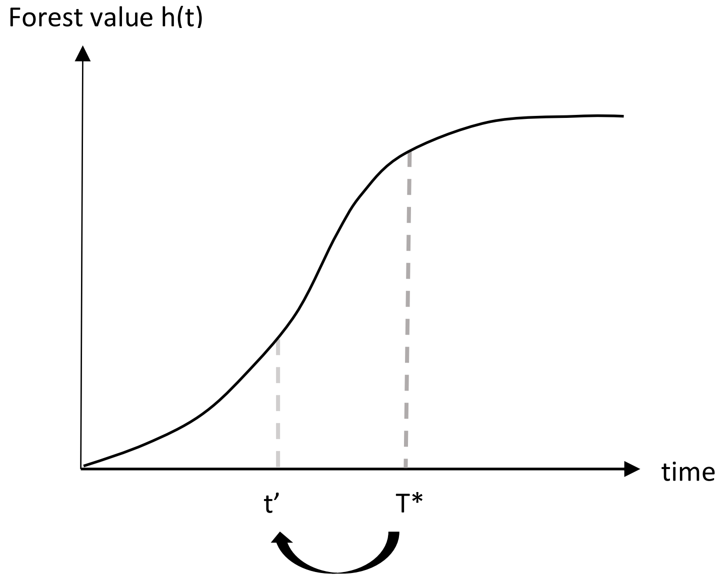

Figure 7. The costs of removing spruces at time t’ versus at time T*.

In rural biotopes, fixed costs include the cost of conservation, 4 987 €/ha, which is based on the value of land. The fixed costs include also the cost of a start-up renovation (thinnings, removing coppice and young trees, and mowing unfavourable vegetation), which is estimated to be 1 862 €/ha. Variable costs, 875 €/ha per year, include the costs of annual grazing and the clearing of coppice if necessary. A guidance cost of 5 % is added to the annual cost. Guidance is needed to make sure that valuable areas are included in the management and less valuable areas are left out. Also, the quality of the management measures is guaranteed with guidance (for instance, the right pressure of pasturing). (Kotiaho et al. 2015, 166.) Rural biotopes are managed for 50 years.

4.5 Estimates for potential supply and demand

The estimates for potential supply and demand are provided in Table 8. I define the potential supply of offsets from each selected habitat type as the area suitable for restoration in Finland. The estimates of these land areas are based on expert assessments and literature (Kemppainen & Lehtomaa 2009; Kotiaho et al. 2015). I estimate demand for offsets drawing on the predictions of future land use change. Tiitu et al. (2015) predict changes in the area of settlements in Finland for a time period of 2013-2040. The data I have utilized includes predictions on the increase of built-up areas (such as residential areas, industrial and commercial complexes, and areas for sports and recreation) as well as infrastructure (including roads, airports, extraction sites, ports and dump sites). The report provides estimates for how many hectares of land in each habitat type will turn into built-up areas or infrastructure. I add the leakage of development impacts outside the area (20 %). I have also taken into account future peatland use (1 000 ha/year), based on a report by Leinonen (2010) and the objectives of Finnish National Energy and Climate Strategy (Kansallinen energia- ja ilmastostrategia 2013).

Direct land-use changes and other activities cause also indirect impacts leading to decrease in habitat quality. Accounting for them will considerably increase the size of land areas under pressure. The magnitude of these impacts is very difficult to estimate. I have included an increase of 100 ha/year in pine mires and 50 ha/year in herb-rich forests and rural biotopes. This is approximately 1 % annual increase to pine mires and up to 50 increase in herb-rich forests and 80 % increase in rural biotopes, as there the land use pressure is very low but due to their higher importance to biodiversity, I assume that their demand is potentially greater when companies purchase indirect offsets.

Table 8. Areas of selected habitat types, areas suitable for restoration and land use pressure, in hectares

	Area of habitat in Finland	**Restorable area**	**Land use pressure**
Pine mires	193 000	193 000	33 000
Herb-rich forests	377 600	264 000	2 500
Rural biotopes	100 000	30 000	3 300

Table 8 shows that potential supply of offsets is strong, especially in pine mires and herb-rich forests. Demand for offsets is considerably lower relative to supply. In particular, predicted land-use pressure on herb-rich forests and rural biotopes is low since the total area of rural biotopes is quite small in Finland, and the share of herb-rich forests of the whole forest land area in Finland (23 000 000 ha) is small as well.

4.6 Uncertainty and Monte Carlo simulations

Recall section 4.3, where uncertainty concerning the success of restoration was examined. Next, I take the estimated uncertainties into account in the model. Monte Carlo simulation allows examining the possible outcomes of habitat restoration when its success may vary. Monte Carlo simulation uses probability distributions to capture the uncertainty of the variables under scrutiny and therefore suits well for the analysis of uncertainty. The simulation was performed in accordance with the results of the survey (combining an expert survey and Monte Carlo simulation, see e.g. Bamber & Aspinall 2013). The results suggest using a triangular probability distribution in the simulation. The minimum, most likely and maximum values are defined – values around the most likely figure are more likely to occur. The program recalculates results, each time using a different value. The values are selected at random from the input probability distribution thousands of times. As a result, it produces distributions of possible outcome values, telling what could happen, and how likely the outcome is.

Variation in the evolvement of restored habitats over 200 years' time span is shown in Figures 8-10. When accounting for uncertainty, the improvement in the state of the restored habitats in rural biotopes and herb-rich forests is still notable, as their state would significantly degrade without restoration. The state of pine mires would improve even without restoration. However, with restoration the improvement is faster and recovery closer to the natural state is more likely.

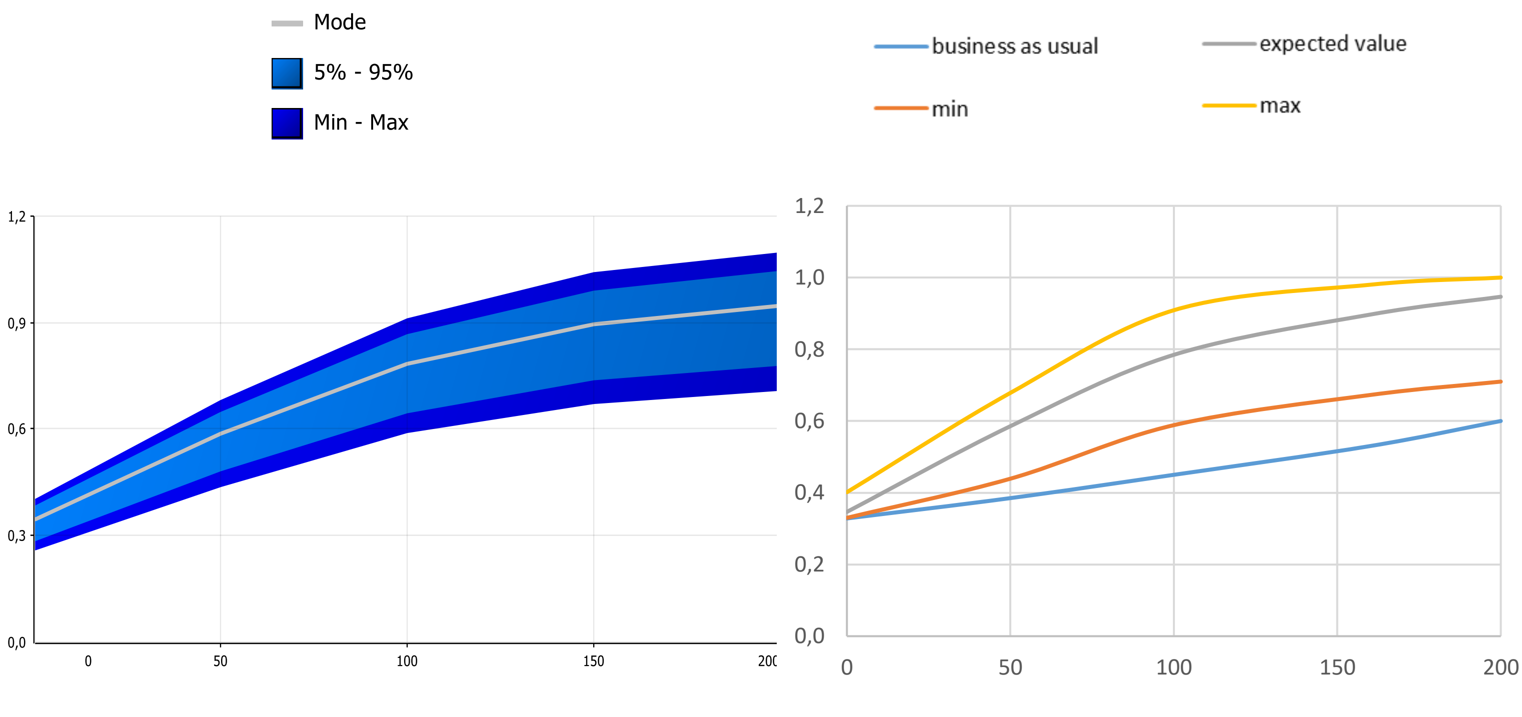

Figure 8. Variation in the evolvement of restored pine mires

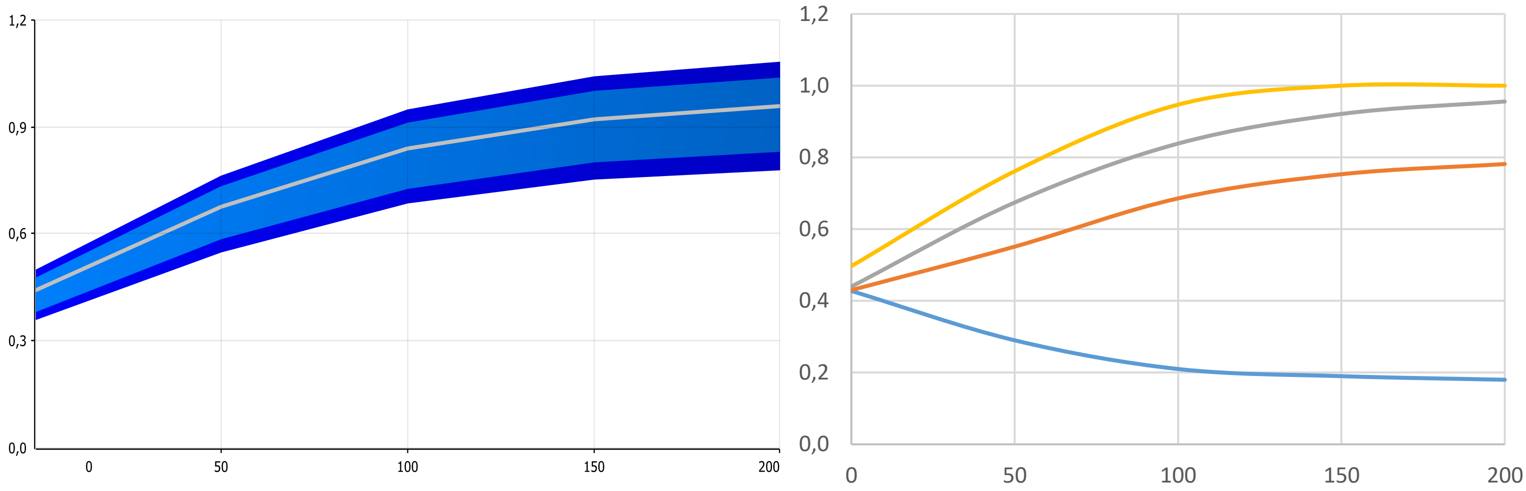

Figure 9. Variation in the evolvement of restored herb-rich forests

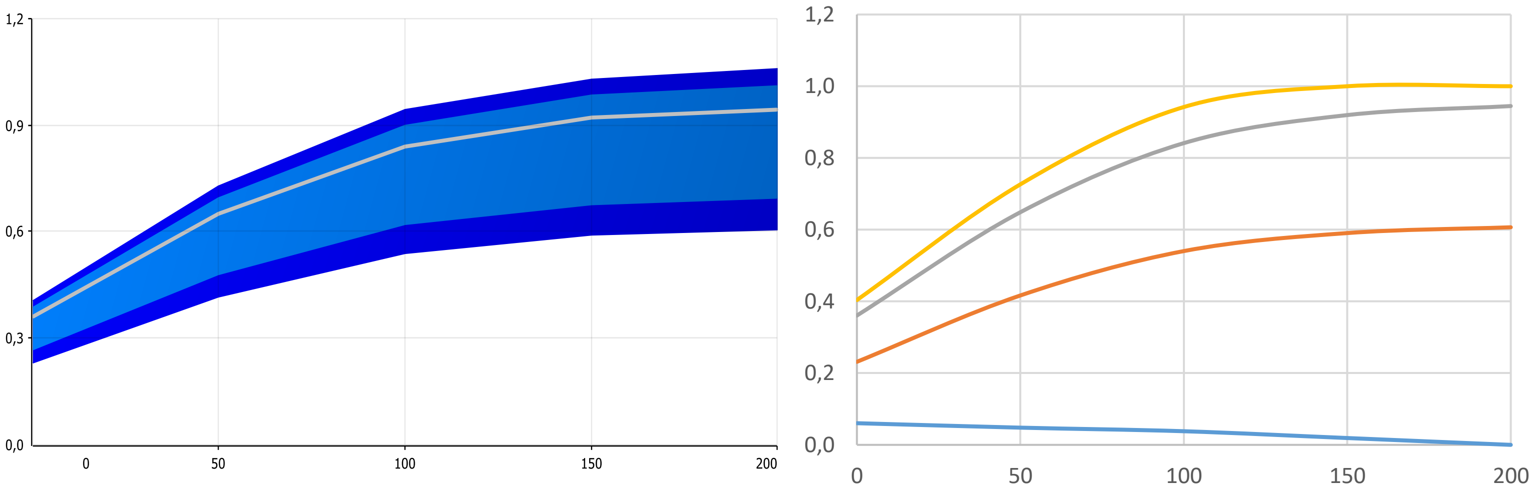

Figure 10. Variation in the evolvement of restored rural biotopes

50 years after the restoration, 90 % of the restoration outcomes are between 0,48–0,65 in pine mires, 0,59–0,73 in herb-rich forests and 0,48–0,69 in rural biotopes. The minimum values are 0,44 in pine mires, 0,55 in herb-rich forests and 0,42 in rural biotopes. Thus, the spread of uncertainty is the widest in rural biotopes. Interestingly, in all selected habitats, the most likely values are closer to upper than lower bounds. The lower bounds are still higher in ecological value than the state without restoration or management measures.

Furthermore, it is useful to know how many restoration sites will fail to provide enough biodiversity gains to be sold as compensation. I assume that if the restoration outcome is less than 90 % of the expected value, restoration has failed. Thus 20 % of the pine mire sites and 10 % of the herb-rich forest sites are not saleable. As the improvement achieved with restoration is so substantial in rural biotopes, I assume that the restoration outcome must be at least 85 % of the expected value to be accepted as compensation. Thus 10 % of the rural biotope sites are not saleable. These figures are quite high and are later taken into account in a risk assessment.

5. Liquid market, with and without time delay

Next, I apply the economic theory to examine the offset markets numerically. Especially, I want to examine how the market equilibrium – prices and quantities traded – depends on trading ratios and the presence of an intermediary. The theoretical analysis was done assuming a liquid market, where mature offsets are available when needed. This type of market may be possible in a case when an intermediary, a broker, works in the market. It helps the demanders and suppliers to meet each other with minimal transaction costs. An intermediary can also safeguard against the risks associated with restoration by buying restored habitats beforehand, so that whenever a degrading of a habitat takes place, the intermediary is able to supply a verified restored habitat for compensation.

I derive the aggregate supply of offsets from each habitat by assuming that fixed costs increase when distance j increases, as remoter sites are more difficult to reach. For simplicity, I assume the sites to be homogenous in other respects, only their accessibility differs. Firms needing offsets face differing trading ratios. Thus, their individual demands differ and are reflected in the aggregate demand curve. Here, market equilibria for rural biotopes represent case 2. Market equilibria for case 3, where the advanced credit release is allowed, is represented in Appendix B.

The benchmark case is provided by assuming a liquid market with an intermediary. Losses and gains in ecological value are ecologically equivalent, and compensation takes place immediately, so the trading ratio is 1 in all habitats. If there are no mature offsets when losses take place, a time delay exists between losses and gains. This must be taken into account by discounting the improvement of ecological value to the present. I assume that it will take 15 years to ensure that habitat restoration has succeeded like expected and offsets mature. A discount rate of 3 % is used here instead of 1 % as in profit functions. I assume that discounting ecological gains requires a higher discount rate. Under these assumptions and using the following equation to match the ecological value of loss and gain: $\sigma = \frac{q^d}{q^s(1+r)^{-t}}$, yields an increase in the trading ratio up to 1,6. Later, the analysis is complicated by assuming that losses and gains are not ecologically equivalent. I examine how that affects the trading ratios and market equilibrium.

5.1 Market equilibrium when gains are ecologically equivalent to losses

Table 9 presents the benchmark case where for each habitat, gains are ecologically equivalent to losses and there is no time delay. Trading ratio equals one, so Table 9 describes the equilibrium of the most liquid market. The results are presented in terms of equilibrium prices, profits, optimal restoration efforts, and total area of compensation sites.

Table 9. Market equilibrium: gains are equivalent to loss, no time delay, trading ratio 1

	price €/ha	**profits €/ha**	**compensation sites in total, ha**
Pine mires	3 315	118	30 321
Herb-rich forests	14 309	431	1 673
Rural biotopes	35 456	408	2 100

Equilibrium prices and quantities vary very much depending on the habitat in question. Offset credits in pine mires are the cheapest and the restored land area is the largest as they require the least costly investment and there the land use pressure is the strongest. Herb-rich forests are three times and rural biotopes ten times more expensive than pine mires. Landowners' average net profits per hectare are slightly over 100 euros for pine mires and around 400 euros per hectare for other habitat types. The costs to companies needing offsets are in total approximately 200 million euros.

I next consider a case where gains are ecologically equivalent to losses but only in the future, after 15 years. Hence, the gains must be discounted to the present. With 15 years' delay and 3 % discount rate, the trading ratio increases from one to 1.6. Features of the market equilibrium are presented in Table 10.

Table 10. Market equilibrium: gains are equivalent to loss, 15 years' time delay, trading ratio 1.6

	price €/ha	**profits €/ha**	**compensation sites in total, ha**
Pine mires	3 486	223	46 729
Herb-rich	14 715	712	2 514
Rural biotopes	35 858	672	3 218

Relative to Table 9, both equilibrium prices and the restored land areas increase for all habitats. A higher trading ratio means that more land is needed to compensate for the biodiversity losses. In terms of land area, the increase is the most dramatic in pine mires. The reason is that restoration increases the ecological value of pine mires quite little and slowly, so much larger areas are needed relative to other habitats. Landowners' profits almost double in pine mires and increase approximately 65 % in other habitats. The costs to companies needing offsets increase to 315 million euros.

5.2 Market equilibrium when gains differ from losses

Next, I consider a case where the biodiversity loss from development is not ecologically equivalent to the gain from restored habitats. Thus, the trading ratio is calculated when the ecological value of gain is less than the loss, as it can be in pine mires and herb-rich forests, or it is more than the loss, which is more likely the case in rural biotopes. In Table 11 are reported the assumed losses and gains and trading ratios both with and without the intermediary (for the impact of the trading ratios on the market equilibrium, recall Figure 4). In Table 11, I set the amount of loss arbitrarily equal to 0,5. The reported gains correspond to the results of the expert survey.

Table 11. Trading ratios when losses and gains in biodiversity are not equivalent (t=15 years, interest rate 3%).

	Loss	**Gain**	**Trading ratio**	
			no time delay	*time delay 15 years*
Pine mires	0,5	0,2	2,5	4
Herb-rich forests	0,5	0,35	1,5	2,3
Rural biotopes	0,5	0,6	0,8	1,3

Table 11 makes it clear that a liquid market with a sufficient stock of mature restored habitats leads to lower trading ratios and expectedly lower market prices. Altogether, required trading ratios are the highest in pine mires as there the increase in ecological value is the lowest. Next, the properties of market equilibrium are examined closer. Picking up the trading ratios from Table 11 leads to the following market equilibrium in Table 12, where biodiversity losses are not ecologically equivalent to gains but there are mature compensation sites available at the time the loss takes place.

Table 12. Market equilibrium: gains are not equivalent to loss, no time delay

	trading ratio	**price €/ha**	**profits €/ha**	**compensation sites in total, ha**
Pine mires	2,5	3 720	369	69 252
Herb-rich	1,5	14 651	667	2 381
Rural biotopes	0,8	35 314	315	1 705

Relative to Table 9, offsets prices from pine mires and herb-rich forests increase due to increased trading ratios. As the trading ratio is the highest in pine mires, their restored land is now almost 2,5 times higher in comparison to the benchmark case. In herb-rich forests, the total area of compensation sites increases moderately (40 %). Landowners' profits behave accordingly: those of pine mires and herb-rich forests increase. The opposite happens to the price of offsets from rural biotopes, as their trading ratio decreases below unity. The state of rural biotopes improves such a considerable amount (see Figure 6) that, unlike in other habitats, it is more likely that gains are higher than losses. The restored land area and profits decrease approximately 20 % in comparison with Table 9.

Next, I add time delay in the previous analysis and employ trading ratios reported in Table 11. The new market equilibrium is presented in Table 13.

Table 13. Market equilibrium: gains are not equivalent to loss, 15 years' time delay

	trading ratio	**price €/ha**	**profits €/ha**	**compensation sites in total, ha**
Pine mire	4	4 061	584	102 041
Herb-rich	2,3	15 131	1 000	3 375
Rural biotopes	1,3	35 661	543	2 671

Again, as trading ratios increase, both prices and the total area of compensation sites increase. Now, the trading ratio in rural biotopes rises also above unity. Due to especially high trading ratios in pine mires, the area of compensation sites now covers almost half of the potential restorable area. The same figure is 1 % in herb-rich forests and 9 % in rural biotopes 9 %.

Table 14 shows the total size of the offset market for each selected habitat, and land areas traded to the year 2040. However, it must be noted that this does not represent the entire offset market in Finland – only the three selected habitat types.

Table 14. The size of the offset credit market, trading ratio 1

	Total size, M€		**Total area, ha**	
Trading ratio	*1*	*1,6*	*1*	*1,6*
Pine mires	101	163	29 109	46 729
Herb-rich forests	24	37	1 673	2 514
Rural biotopes	74	115	2 100	3 218
TOTAL	**199**	**315**	**32 882**	**52 461**

When trading ratios are equal to one and there is no time delay, the market size is estimated to be 200 million euros in total. Approximately 33 000 hectares of land will be restored and conserved. If trading ratios increase to 1,6, the total size of the market is 315 million euros and over 52 000 hectares.

5.3 Sensitivity analysis

In the previous section, I assumed that when gains achieved with restoration are not ecologically equivalent to biodiversity losses from development, the improvement in ecological value is measured at point in time $\tau = 50$. Below, I examine how changing τ affects trading ratios.

Table 15. The impact of τ to trading ratios, no time delay

	Loss	Gain (τ=25)	Gain (τ=50)	Gain (τ=100)	σ (τ = 25)	σ (τ = 50)	σ (τ = 100)
Pine mires	0,5	0,1	0,2	0,3	5	2,5	1,7
Herb-rich forests	0,5	0,2	0,35	0,6	2,5	1,5	0,8
Rural biotopes	0,5	0,45	0,6	0,8	1,1	0,8	0,6

Table 15 shows that the later the gains are calculated, the higher the gains are. The reason is that as time goes by, the difference between a restored habitat and a habitat in a business-as-usual scenario grows. Thus, trading ratios are higher if the gains are calculated at an earlier point in time. When trading ratios are higher, equilibrium prices and profits for landowners as well as the restored land areas increase. Table 16 shows how changing τ to 25 affects the market equilibrium.

Table 16. Market equilibrium: τ = 25, no time delay.

	trading ratio	price €/ha	profits €/ha	compensation sites in total, ha
Pine mires	5	4 259	710	121 065
Herb-rich forests	2,5	15 240	1 075	3 600
Rural biotopes	1,1	35 525	453	2 293

We see that relative to Table 12, trading ratios increase, and the increase is especially high in pine mires. The equilibrium price increases 13 %, profits increase 90 % and the area of compensation sites increases 75 %. In herb-rich forests, the equilibrium price increases 4 %, and the increase in profits and the area of compensation sites is 60 % and 50 %. In rural biotopes, the price increases less than 1 %, but the increase in profits and the area of compensation sites is 44 % and 34 %. Thus, the determination of τ has a significant impact and must be noted when interpreting the results of this thesis.

6. The role of an intermediary: pricing transaction costs and the risk of failure

The intermediary can reduce the transaction costs of market participants by providing broker services. It can also play a constructive role in reducing the risk of failure in the market. In the previous case of liquid markets, these services were assumed to be costless but naturally, this is not plausible. In this chapter, I consider how additional fees collected by the intermediary affect the market equilibrium. First, the intermediary collects a payment as a fee from the services it provides to reduce market participants' transaction costs. This is added to the offset price. Second, the intermediary estimates the share of failed projects and prices the risk in the broker services, which again shows up in offset prices.

Figure 11 qualitatively illustrates the impacts of the additional fees. Pricing the transaction costs and risks affects like a tax: if levied on buyers the (after-premium) demand curve shifts downwards, and if levied on suppliers the (after-premium) supply curve shifts upwards. In both cases, the price increases to p^{**} and the fee collected by the intermediary is an amount represented by area $p^{**}ABp_s$. Due to higher prices, the area of compensation sites in total will decrease from Q^* to Q^{**}.

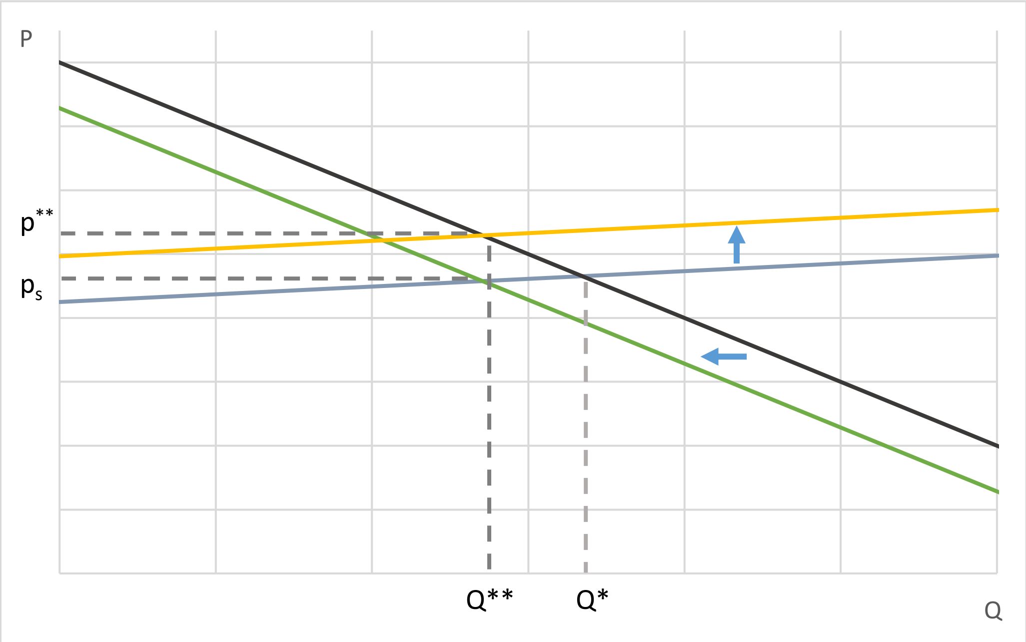

Figure 11. The effect of additional fees to the market.

6.1 Transaction costs

In offset markets, transaction costs can incur when the developer must learn about offset requirements, negotiate requirements with the regulator, find suppliers and negotiate contracts with the suppliers. Transaction costs to the supplier can include the costs of learning about offsets and what can be supplied, negotiating contracts with buyers and the regulator, monitoring and reporting compensation measures and responding to enforcement measures in case the compensation sites do not meet their requirements. The intermediary can reduce these costs by providing information, broker services and so on. (Coggan et al. 2013a.)

Now, I assume that the intermediary includes an additional payment in the price as a fee from the services it provides to reduce market participants' transaction costs. In pine mires, the fee is a 10 % addition to the price and because of the higher prices, in herb-rich forests and rural biotopes, the fee is 5 % of the offset price. The new market equilibrium with the added fee is presented in Table 17.

Table 17. Market equilibrium with an intermediary fee, trading ratio 1

	transaction cost, €/ha	**buyer price, €/ha**	**profits, €/ha**	**compensation sites in total, ha**
Pine mires	332	3 626	105	28 311
Herb-rich forests	715	14 947	317	1 513
Rural biotopes	1 773	37 099	323	1 738

Relative to Table 9, buyer prices increase 9 % in pine mires and 5 % in herb-rich forests and rural biotopes. In pine mires and herb-rich forests, both the decrease in profits and in the total area of compensation sites is approximately 10 %. Because of the high offset prices in rural biotopes, a fee with 5 % share has a bigger impact on the market equilibrium: profits and the total area of compensation sites are now approximately 20 % lower.

6.2 Risk premium

It is likely that all restoration projects do not succeed. This creates a risk to the market and nature: buyers buy compensations which do not improve the state of habitats. The intermediary can play a constructive role in reducing the risk of failures in the market. In the previous case of liquid markets, it is implicitly assumed that the intermediary safeguards against failed compensations. The intermediary can price the economic and ecological risks in the broker services by estimating the monetary value of failed projects. This risk premium or insurance is assumed to include also the fee calculated earlier.

Monte Carlo simulation results provide data on how many percent of the restored sites will not recover as expected. When restoration is not successful, the sites are not saleable. The intermediary calculates the revenue loss from failures and allocates a risk premium in the market. I assume that if the outcome of restoration is less than 90 % of the expected value, restoration has failed and there is no compensation to be sold. For instance, in pine mires, this means that 20 % of the restored area, 6 604 hectares, is useless and the loss is worth approximately 20 million euros. Per hectare the loss is 663 euros, which is the risk premium collected by the intermediary. In herb-rich forests 10 % of the sites are not saleable. As the improvement achieved with restoration is so substantial in rural biotopes, I assume that sites that achieve at least 85 % of the expected value are saleable as compensation. Thus 10 % of the sites are not saleable. The new market equilibrium with risk premiums (and trading ratio equal to one) is presented in Table 18.

Table 18. Market equilibrium with risk premium

	risk premium €/ha	buyer price €/ha	profits €/ha	compensation sites in total, ha
Pine mires	663	3 937	92	26 300
Herb-rich forests	1 431	15 585	324	1 354
Rural biotopes	7 091	42 027	69	654

Comparing Table 18 with Table 9 reveals that prices are now approximately 10-20 % higher and restored land areas in pine mires and herb-rich forests approximately 20 % lower. In rural biotopes, the compensated land areas are 70 % lower which is a considerable decrease. Thus, the impact of the elimination of ecological risks is not especially large in the market, except for the land area in rural biotopes. In contrast, its impact on biodiversity is considerable. Economic risk is the greatest in rural biotopes and the risk in terms of land area is the greatest in pine mires.

Finally, I consider how increasing the risk premium affects the market equilibrium. Above, I assumed that if the restoration outcome is less than 90 % of the expected value, or 85 % in rural biotopes, restoration has failed and there is no compensation to be sold. In Table 19 are provided results on how increasing this requirement by 5 % will affect risk premiums and the market equilibrium.

Table 19. Market equilibrium with higher risk premiums

	risk premium €/ha	buyer price €/ha	profits €/ha	compensation sites in total, ha
Pine mires	1 160	4 403	73	23 285
Herb-rich forests	3 577	17 500	164	875
Rural biotopes	10 637	45 312	-99	-69

Relative to Table 18, we see that the impact is substantial: risk premiums increase by 50-75 % in pine mires and rural biotopes, and it is 2,5 times higher in herb-rich forests. Thus, buyer prices increase and profits decrease considerably. As offsets are expensive in rural biotopes, risk premiums are very high and profits are now negative. Thus, it is not feasible to raise the level of the risk premiums excessively high.

7. Conclusions and discussion

Internationally, restoring ecosystems has become an important way to slow down the loss of biodiversity and maintain ecosystem services. In this thesis, I have developed an equilibrium model to study biodiversity offset markets and applied the analytical model to three selected habitats. I have examined how trading ratios, the presence of an intermediary and the realisation of risks associated to uncertainty affect the market equilibrium: offset prices and quantities traded.

The results show that the size of offset markets could potentially be considerable, and providing offsets could be a profitable business for landowners. There is enough land and suitable habitats for compensations in Finland, even if trading ratios are relatively high. In habitats where restoration or nature management is laborious and expensive, offset prices are high, and especially when continuous management is required, compensation can be very costly. The ecological equivalence and possible time delay between biodiversity losses and gains impact trading ratios and thus, have a major impact on the market equilibrium. An intermediary that provides broker, offset aggregator and/or banker services may significantly decrease the costs of compensation for developers.

In Chapter 3, I considered how allowing the sale of a share of the credits upfront will affect the optimal restoration effort. In Appendix B, a closer analysis can be found on the market equilibrium in rural biotopes when advanced credit release is allowed. Comparing two cases of rural biotopes shows that irrespective of trading ratios, with advanced credit release, the prices are lower and both profits for landowners and land areas for compensation are higher. However, the difference between the two cases is quite modest. The results are consistent with BenDor et al. (2011) who did not found a significant link between advanced credit release policies and actual number of banks and credit production in the wetland mitigation market in the US. Market entry is primarily related to regional geography (the prevalence of suitable habitats for compensation) and regional economic growth (i.e., demand for offsets). As the advanced credit release policy increases ecological risks, its usefulness can be debated in the light of these results. If the advanced credit release rate is set, a balance must be found between landowners' participation and incentives that induce landowners to invest effort in creating a successful compensation after the initial credit release (BenDor et al. 2014).

The shortcomings of the existing offsetting schemes and the extensive scientific literature have been considered developing the model, in order to study an advanced, viable offsetting mechanism and feasible offset markets. Using a banking mechanism is the first step, as it is seen as the most developed mechanism to implement offsetting (Briggs et al. 2009). However, anticipatory approaches, such as habitat banking, can create some disincentives (McKenney & Kiesecker 2010). They require offset suppliers to foresee project impacts before they have occurred, which may bear risks. The suppliers can also suffer from substantial upfront costs if there is no ability to receive income by advanced credit release. By adding the intermediary to the market, the disincentives may be diminished.

How likely it is to have an intermediary participating in the market? Intermediaries have been identified in many existing offsetting schemes (Coggan et al. 2013a; OECD 2016, 176-192). They can be deliberately created for a market by a regulator or they may emerge privately (Coggan et al. 2013a). By creating an intermediary, the regulator aims to enhance the number and quality of trades, whereas private brokers can profit from bringing specialist knowledge to market participants and reducing their transaction costs by collecting a fee. Both ecological and economic risks may decrease as the intermediary providing banker services guarantees that offset credits are mature at the time of sale and safeguards against failures in restoration by guaranteeing that all offsets provide good quality. The results show that as long as the fees and risk premiums collected by the intermediary are not excessively high, the impact of pricing these services in the market is quite modest, apart from rural biotopes where the trades decrease considerably.

Trading ratios calculated in the model are relatively low in comparison with those found in literature (Gibbons et al. 2015; Laitila et al. 2014; Moilanen et al. 2009). One reason is that only lower bounds of uncertainty are included in the calculation of the trading ratio. I did not consider the possibility that restoration fails completely – this scenario was included in the risk premium as the intermediary bears the risk of failure. Secondly, there are also other sources of uncertainty which would increase the trading ratio. They can be taken into account by adding an error weight (Moilanen et al. 2009). Small error weights can be used if there is a lot of experience and knowledge regarding the restoration and management of the habitats studied and the site is well surveyed. A higher error is needed if an area is poorly surveyed or there is lack of knowledge, for instance, if a new restoration technique is tested. Trading ratio increases substantially if it is assumed that success between distinct restoration sites is correlated to some degree. However, the feasibility of very high trading ratios (increasing from dozens to hundreds) is debatable. Trading ratios employed here are consistent with the ones used in practice (Bull et al. 2016), except for the fact that proposed ratios are rarely below 1.0.

There are some limitations in the model and the application. A major challenge was that this kind of analytical modelling on the offset markets had not yet been made, and the data available on the existing offset markets, realized costs and prices is very limited. In order to estimate supply and demand, I had to rely on expert assessments, and apply and combine information from many documented sources. Also, many assumptions had to be made in order to include important factors in the study without any support from similar analyses or experiences from the existing offset markets.

Determining τ, the point in time when the improvement in the ecological state of the habitat is calculated, has a significant impact on trading ratios and thus, equilibrium prices, profits and traded compensation sites. It must be noted when interpreting the results. In the sensitivity analysis, I compared a few alternatives (25, 50 and 100 years) and the differences in the equilibrium prices and traded land areas were substantial. However, there is no unambiguous answer to what the level of τ should be. Expert assessments may be the best way to make certain that τ is set to a point in time which is low enough to be feasible but high enough to ensure that the calculation of ecological gains is reliable. The same applies to the level of intermediary fees and risk premiums. As there is no data on the level of these kind of payments from existing offset markets, I chose a few alternatives. The level of the payments is a matter of agreement. Thus, the results concerning the impact of risk premiums and intermediary fees should only be used to analyse the impact on the market in general.

Here compensation is assumed to be compulsory – all adverse impacts on biodiversity from land-use change must be offset. If offsetting is voluntary, it will strongly affect demand and the market size will shrink. I have assumed trading in-kind, but if trading up is possible, purchasing credits from rural biotopes and herb-rich forests could increase, as they are more valuable to biodiversity than mires. However, it is likely that the high offset prices limit trading up. Trading down is not preferred, but if it was allowed, there would be a risk that demand would channel predominantly to pine mires as they are three to ten times cheaper than other habitats.

The expert survey received some feedback and criticism. A few concepts I used in the survey were questioned: especially terms natural state and restoration are controversial in rural biotopes and herb-rich forests. Thus, I use the term target state instead of natural state. Also, the concept of restoration is not entirely suitable for these habitats – nature management is the correct term. Some respondents were worried that because of the controversial use of these terms, the questions of the survey might have been misunderstood. The results suggest otherwise, as the standard deviations of the answers were low. The use of ELITE report was also criticized by some respondents. I have considered the shortcomings of the report and for instance, cost calculations have been adjusted accordingly.

Limiting the example of mire restoration to only pine mires does not necessarily correspond to reality. Mires usually form a mosaic pattern, consisting of different mire types, minerotrophic parts and ombrotrophic parts. This mosaic as a whole is valuable for biodiversity and more challenging to restore. Restoring ombrotrophic pine mires is rather simple but assessing benefits for biodiversity is a more complex issue, as separate pine mire patches are not particularly valuable. However, for the purposes of this study, a simpler example of mire restoration was more suitable. Modelling the restoration of large combinations of different mire habitats is out of the scope of this thesis.

This thesis has aimed to provide a new kind of analysis of the biodiversity offset markets on the market level. The analytical model introduced here could be used to further study the different factors in the market: taking a closer look on trading ratios, adding carbon offsets in the market, and so on. The model could be further developed and tested with a case study if data on realized offset trades would become available. As literature on intermediaries in offset markets is limited to a few case studies, closer analysis is needed, for instance, on the different roles of the intermediary and how the intermediaries impact transaction costs and prices, and ease trades in the market.

Implementing an offsetting mechanism would improve the current state of biodiversity and habitat restoration in Finland. There is a lot of experience and knowledge regarding the restoration and management of mire and forest habitats in Finland (Aapala et al. 2013; Similä & Junninen 2011) as well as degraded habitats suitable for restoration and management, which is an advantage. The results show that there is potential to both supply of and demand for offsets in Finland. Still, ecological compensations alone will not be enough. Preserving the most valuable species and habitats is essential, and all impacts cannot be compensated. Irreplaceable, extremely vulnerable ecosystems and habitats or endangered species are always no-go areas where offsetting cannot be applied. Nevertheless, ecological compensations can potentially be an important addition to the policy mix, in order to halt the alarming rate of biodiversity loss, and ensure the well-functioning ecosystem services also in the future.

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Appendix A

Comparative statics: partial derivatives in respect to optimal effort

$$\frac{\partial x^*}{\partial \alpha} = \frac{1}{2\beta}$$

$$\frac{\partial x^*}{\partial \beta} = -\frac{\alpha}{2\beta^2} + \frac{w}{p2\beta^2\emptyset\frac{1}{1-e^{-k(t-l_0)}}}$$

$$\frac{\partial x^*}{\partial w} = -\frac{1}{p2\beta\emptyset\frac{1}{1-e^{-k(t-l_0)}}}$$

$$\frac{\partial x^*}{\partial p} = \frac{w}{p^2 2\beta\emptyset\frac{1}{1-e^{-k(t-l_0)}}}$$

$$\frac{\partial x^*}{\partial \emptyset} = \frac{w}{p2\beta\emptyset^2\frac{1}{1-e^{-k(t-l_0)}}}$$

$$\frac{\partial x^*}{\partial \tau} = \frac{ke^{-k(\tau-l_0)}w}{p2\beta\emptyset}$$

$$\frac{\partial x^*}{\partial k} = \frac{w(\tau-l_0)e^{-k(\tau-l_0)}}{p2\beta\emptyset}$$

$$\frac{\partial x^*}{\partial l_0} = \frac{-ke^{-k(\tau-l_0)}w}{p2\beta\emptyset}$$

Case 2

$$\frac{\partial x^*}{\partial r} = -\frac{w(n-m)(1+r)^{-m-1}}{p2\beta\emptyset\frac{1}{1-e^{-k(t-l_0)}}(1+r)^{-n}}$$

$if\ m > n$

$$\frac{\partial x^*}{\partial r} > 0$$

$if\ m < n$

$$\frac{\partial x^*}{\partial r} < 0$$

Case 2, annuity

$$\frac{\partial x^*}{\partial r} = \frac{m(1+r)^{-m-1} + n(1+r)^{-n-1}[1-(1+r)^{-m}]}{(1+r)^{-n}}$$

Case 3

$$\frac{\partial x^*}{\partial r} = -\frac{wm(1+r)^{-m-1}[Br\theta + Br(1-\theta)(1+r)^{-m}] - [B\theta + B(1-\theta)(1+r)^{-n} - Bnr(1-\theta)(1+r)^{-n-1}][w(1-(1+r)^{-m})]}{[Br\theta + Br(1-\theta)(1+r)^{-n}]^2}$$

$$\frac{\partial x^*}{\partial \theta} = \frac{w[1-(1+r)^{-m}]}{[\theta + (1-\theta)]^2(1+r)^{-n}}$$

Appendix B

Advanced credit release in rural biotopes

In Chapter 3.2, there are three cases representing differences in timing of costs and revenue to offset suppliers. In the third case, advanced credit release, the costs of nature management occur annually and a part of unrealized offsets can be sold immediately, and the rest can be sold at a future point of time, $n = 5$. Table 20 shows the market equilibria when advanced credit release is allowed in rural biotopes. The advanced credit release rate is 0,5, i.e. 50 % of the offset credits are saleable immediately after regular nature management has begun.

Table 20. Market equilibria for rural biotopes when advanced credit release is allowed

	Trading ratio	price €/ha	profits €/ha	compensation sites in total, ha
Gains = loss, no delay	1	34 614	435	2 285
Gains = loss, 15 yrs delay	1,6	35 029	714	3 510
Gains ≠ loss, no delay	0,8	34 468	334	1 854
Gains ≠ loss, τ = 25, no delay	1,1	34 685	483	2 496
Gains ≠ loss, 15 yrs delay	1,3	34 826	577	2 910
Risk premium 6 923 €/ha	1	41 057	114	868
Risk premium 10 384 €/ha	1	44 278	-45	159
Intermediary fee 1 731 €/ha	1	36 225	354	1931

Relative to the results in Chapters 5 and 6, where all credits from rural biotopes are saleable after 5 years, equilibrium prices are 2 % lower, profits are 6 % higher and total area of compensation sites is 9 % higher. Thus, the impact of the advanced credit release is quite modest. Increasing the advanced credit release rate or increasing n in the case where all credits are sold at n would increase the difference between these two alternatives.

Appendix C

The survey (in Finnish)

Kyselyiden johdannot

Kysely rämeiden ennallistamisen epävarmuudesta

Tarkastelun kohteena ovat ojitetut karuhkot rämeet Suomessa. Elinympäristön tilan vaihtelu on määritelty välille 0–1, missä 1 kuvastaa täysin luonnontilaista elinympäristöä ja 0 täysin luonnontilaisesta muuttunutta. Rämeiden tilan arviointi perustuu vesitalouden keskimääräiseen tilaan ja puuston keskimääräiseen määrään.
– Vesitalouden tila on lähtötilassa 30, luonnontilassa 100.
– Puustoa on lähtötilassa 30 m3/ha, luonnontilassa 20 m3/ha.
– Tämän perusteella rämeiden lähtötilaksi on arvioitu keskimäärin 0,32.

Ennallistamista ovat ne toimenpiteet, joilla edistetään heikentyneen ekosysteemin palautumista kohti luonnontilaa. Kohdealueen sisällä tehdyt toimet riittävät keskimäärin mahdollistamaan vesitalouden ja elinympäristön palautumisen. Ennallistamiseen kuuluu myös suojelualueen perustaminen.

Ennallistamistoimenpiteitä ovat ojien tukkiminen ja puuston poisto. Ojia tukkimalla nostetaan suoveden pinta luontaiselle tasolle ja palautetaan vesien virtailu suolla luontaisille reiteilleen. Puuston poisto palauttaa suon avoimuuden ja vaikuttaa vedenpinnan tasoon. Puustoa poistetaan niin paljon, että puuston määrä on luonnontilaisen kaltainen.

Ilmastonmuutos tulee vaikuttamaan elinympäristöjen tilaan, mutta näissä arvioissa ilmastonmuutoksen vaikutus sivuutetaan.

Ilmoita arvioimasi todennäköisyydet prosentteina välillä 0-100 %.

Kysely perinnebiotooppien ennallistamisen epävarmuudesta

Perinnebiotooppeja ovat erilaiset laidunnuksen tai niittämisen tuloksena syntyneet avoimet kedot, tuoreet ja kosteat niityt sekä hakamaat ja lehdesniityt. Tarkastelu kohdistuu perinnebiotooppeihin Suomessa.

Elinympäristön tilan vaihtelu on määritelty välille 0–1, missä 1 kuvastaa luonnontilaista elinympäristöä ja 0 täysin luonnontilaisesta muuttunutta. Perinnebiotooppien tilan keskimääräistä heikentymistä kuvaavat kasvillisuuden rakenne, alueen avoimuus ja maan muokkaamattomuus. Heikennystä vertaillaan tilaan, jossa kaikki tekijät ovat 100-prosenttisessa kunnossa. Tässä kyselyssä termi "luonnontilainen" eli arvo 1 viittaa tähän tilaan, jossa perinnebiotooppi on 100-prosenttisessa kunnossa. Kasvillisuuden rakenne lähtötilassa on 10 %, avoimuus 20 % ja muokkaamaton maa 60 %. Tämän perusteella perinnebiotooppien lähtötilaksi on arvioitu keskimäärin 0,06.

Ennallistamiseksi määrittelemme ne toimenpiteet, joilla edistetään heikentyneen ekosysteemin palautumista kohti tilaa ennen heikennystä. Ennallistamiseen kuuluu myös suojelualueen perustaminen. Ennallistamistoimenpiteitä tässä tapauksessa ovat peruskunnostus ja laidunnus. Peruskunnostus käsittää alueen puuston harvennusta lehtipuita ja ylispuita ja/tai maisemapuita suosien, vesakon tai nuoren puuston raivausta tai harvennusta, sekä avoimilla alueilla epäsuotuisan kasvillisuuden niittoa. Perinnebiotooppien luontotyyppien ja lajiston ylläpitämiseksi toteutetaan vuosittain toistuvaa laidunnusta. Se sisältää tarvittaessa ylläpitoraivauksen. Sopiva laidunnuspaine valitaan tapauskohtaisesti.

Ilmastonmuutos tulee vaikuttamaan elinympäristöjen tilaan, mutta näissä arvioissa ilmastonmuutoksen vaikutus sivuutetaan.

Ilmoita arvioimasi todennäköisyydet prosentteina välillä 0-100 %.

Tarkastelun kohteena ovat lehdot Suomessa. Lehtojen tilan keskimääräistä heikentymistä kuvaavat järeiden (läpimitaltaan yli 40 cm) puiden väheneminen, lahopuun määrän väheneminen sekä lehtipuun määrän väheneminen. Elinympäristön tilan vaihtelu on määritelty välille 0–1, missä 1 kuvastaa luonnontilaista elinympäristöä ja 0 täysin luonnontilaisesta muuttunutta.

– Järeän puun määrä lähtötilassa 10,1 kpl/ha, luonnontilassa 30 kpl/ha.
– Lahopuun määrä lähtötilassa 7,0 m3/ha, luonnontilassa 100 m3/ha.
– Lehtipuun määrä lähtötilassa 92,0 m3/ha, luonnontilassa 100 m3/ha.
– Tämän perusteella lehtojen lähtötilaksi on arvioitu keskimäärin 0,44.

Ennallistamiseksi määrittelemme ne toimenpiteet, joilla edistetään heikentyneen ekosysteemin palautumista kohti luonnontilaa. Ennallistamiseen kuuluu myös suojelualueen perustaminen. Ennallistamisen toimenpiteet käsittävät istutuskuusten poistamista tai vähentämistä sekä luontaisen kuusettumisen estämistä. Myös lahopuita voidaan tuottaa sopivissa paikoissa. Toimenpiteet toistetaan 10–20 vuoden välein. Lehtojen hoidon tavoitteena on lehtipuuvaltaisuus ja puustorakenteen monipuolisuus.

Ilmastonmuutos tulee vaikuttamaan elinympäristöjen tilaan, mutta näissä arvioissa ilmastonmuutoksen vaikutus sivuutetaan.

Ilmoita arvioimasi todennäköisyydet prosentteina välillä 0-100 %.

Kysymykset (samat kaikissa kolmessa kyselyssä)

1. Elinympäristön lähtötila on X. Elinympäristöä ei suojella eikä siellä toteuteta ennallistamistoimia. Kuinka lähellä luonnontilaa arvioit elinympäristön olevan, välillä 0-1 (0 = täysin muuttunut luonnontilaisesta, 1 = täysin luonnontilainen)

a. 100 vuoden kuluttua?
b. 200 vuoden kuluttua?

Kuinka varma olet vastauksistasi (1 = täysin epävarma, 5 = täysin varma)?

2. Elinympäristön lähtötila on X. Elinympäristö suojellaan sekä ennallistetaan johdannossa mainituilla toimenpiteillä. Kuinka lähellä luonnontilaa arvioit elinympäristön olevan 100 vuoden kuluttua? Koska ennallistamisen lopputulos vaihtelee antamasi arvon ympärillä, arvioi lisäksi elinympäristön tilan uskottavaa ylärajaa ja alarajaa.

Millä todennäköisyydellä palautuminen toteutuu (välillä 0-100 %)?

Kuinka varma olet vastauksistasi (1 = täysin epävarma, 5 = täysin varma)?

3. Muutetaan tarkastelukulmaa. Arvioi nyt, missä ajassa ja millä todennäköisyydellä elinympäristö saavuttaa määrätyn tilan.

Kuinka monta vuotta vie, että elinympäristö edellä mainituilla ennallistamistoimenpiteillä saavuttaa tilan X?

Millä todennäköisyydellä tämä toteutuu (välillä 0-100 %)?

Kuinka varma olet vastauksistasi (1 = täysin epävarma, 5 = täysin varma)?

4. Lopuksi arvioi ennallistamisen vaikutuksia pitkällä aikavälillä.

Millä todennäköisyydellä elinympäristö palautuu luonnontilaan 200 vuodessa johdannossa mainituilla ennallistamistoimilla (välillä 0-100 %)?

Kuinka varma olet vastauksestasi (1 = täysin epävarma, 5 = täysin varma)?